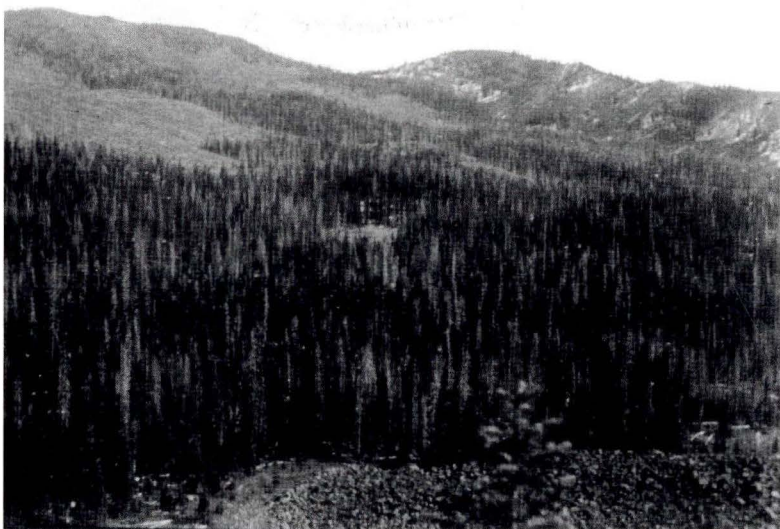# Historical Range of Variability for Forest Vegetation of the Grand Mesa National Forest, Colorado
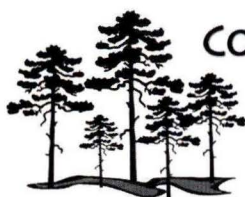
Dominik Kulakowski
Thomas T. Veblen
Department of Geograpny
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Photo: G.B. Sudworth, 1898

USDA

FOREST SERVICE
U S
DEPARTMENT OF AGRICULTURE

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www.cfri.colostate.edu

Colorado State University

1900 108°

—39°

2000 108°

—39°

Forest cover (1998)
■ Aspen
■ Spruce - Fir
■ Grass / Shrub

Kilometers
0 5 10 20

N

HISTORICAL RANGE OF VARIABILITY FOR FOREST

VEGETATION OF THE GRAND MESA NATIONAL FOREST,

COLORADO

September, 2006

Dominik Kulakowski
Thomas T. Veblen
Department of Geography, University of Colorado, Boulder, CO

USDA
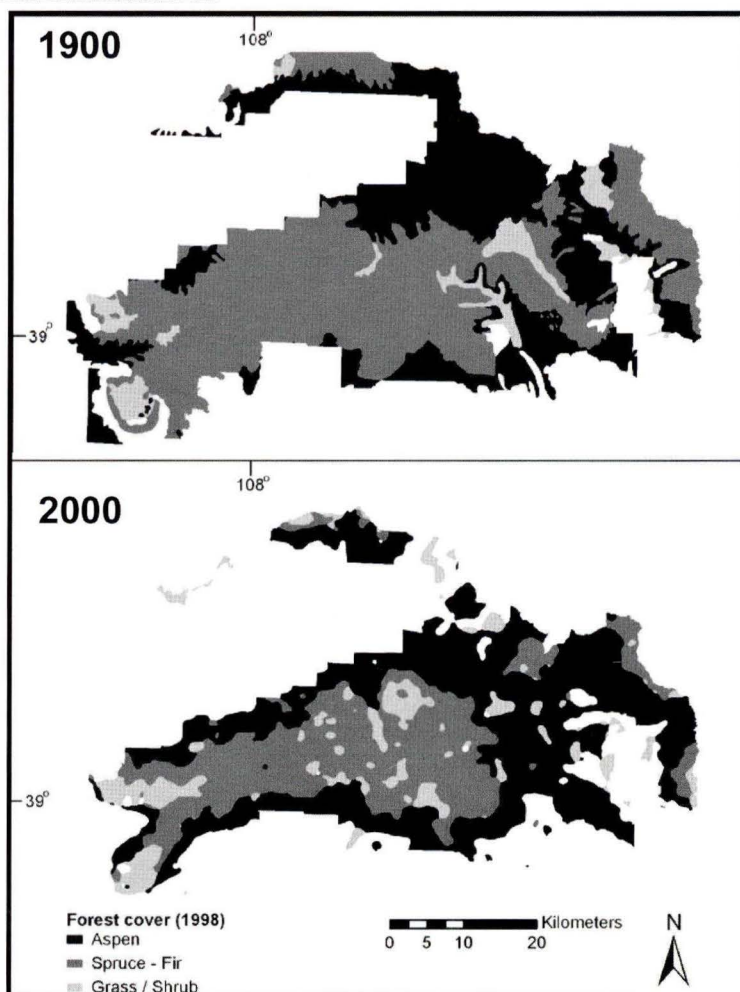
USDA Forest Service
Rocky Mountain Region
740 Simms St
Golden, CO 80401

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Preface and Acknowledgements

This assessment of the historic range of variability (HRV) of the forest vegetation of Grand Mesa National Forest has been motivated by the widespread recognition that forest resource planning and decision-making would benefit from a series of assessments of the HRV of the ecosystems which comprise the National Forest lands of the Rocky Mountain region. This report was developed under a Cooperative Agreement between the Regional Office of the Forest Service and the University of Colorado initiated in 1999. It is part of a program to provide HRV reports on all the National Forests in Region 2 under the leadership of Claudia Regan, Regional Ecologist for Region 2. This report has benefited from her insights into both the science and management aspects of the HRV process. Her leadership and perseverance in this long process are deeply appreciated.

While the interpretations in this report are the responsibility of its authors, we have endeavored to address critiques and alternative interpretations from a wide variety of sources. Over the past five years we have benefited from numerous discussions with many Forest Service personnel of topics directly related to this HRV assessment. In the current report we have addressed issues and alternative interpretations which have become apparent to us from these discussions with individuals and in numerous workshops. We also have addressed issues which were raised by Forest Service personnel in written reviews of earlier drafts of the report. Likewise, the current report addresses and has benefited from written reviews of earlier drafts of this report from researchers in the Region and from five anonymous reviewers selected by the Ecological Society of America. The authors of this report particularly want to acknowledge the detailed comments and discussion provided by numerous staff of Grand Mesa National Forest during a one-day meeting with us to discuss the initial draft of the report. We feel that the report benefited significantly from the critiques provided at that meeting and we made substantive changes in the draft to address those critiques. Nevertheless, there remain issues on which our interpretations may differ from those of the staff present at that meeting. In the final version of the report we have identified where there are important uncertainties or needs for further research that potentially could resolve those issues.

Finally, we acknowledge that some of the background material, especially in chapters 1 and 2, has only been slightly modified from text published in Veblen and Donnegan (2006). We apologize for the redundancy to readers of both reports, but we felt that some redundancy was justified so that this report could be interpreted as a stand alone document that would be of maximum benefit to readers interested in the respective National Forests.

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- Numerous personnel of the Grand Mesa, Uncompahgre, Gunnison National Forest who wish to remain anonymous

Contents

1. Introduction

In recent decades, an emphasis has been placed on ecosystem sustainability and integrity in the utilization of National Forest lands and other public lands (Kessler *et al.* 1992, De Leo and Levin 1997). In addition to providing products and services, the climate interactions and carbon sequestration provided by national forests are seen as being critical to global environmental security. Therefore, sustaining ecosystems for present and future generations is the foremost responsibility of resource managers. The focus of forest management is placed on sustaining forest ecosystems, which are part of a larger functioning ecosystem, rather than just sustaining yields of resources. At the core of ecosystem sustainability is the maintenance of ecological integrity, which rests in the maintenance of a given ecosystem's physical environment, its biota (including its community structure and composition), and ecosystem processes. Ecological integrity may well be necessary for an ecosystem to continue to support services of value to human society (De Leo and Levin 1997). To assess ecosystem integrity, it is important to characterize the structure and function of an ecosystem and to be able to perceive the departure from historical conditions as a result of human activity (De Leo and Levin 1997). An understanding of how ecosystems functioned and sustained themselves in the absence of human modification of ecological patterns and processes provides a concrete model of ecosystem sustainability. The emphasis in on-the-ground management can be placed on utilizing forests in accord with processes that maintain and have maintained ecosystem integrity and sustainability. An associated understanding of both the structural and the functional components of such a model is critical to successful management.

Ecosystem management (Christensen et al. 1996) has been defined as: "Management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem structure and function." An important component of the modern ecosystem management paradigm is explicit recognition of the dynamic character of ecosystems. Ecosystem management is not intended to provide maintenance of any *status quo* in ecosystem conditions but rather accepts that change is an inherent characteristic of ecosystems across both space and time.

Under the paradigms of ecological integrity, sustainability, and ecosystem management, it is important for resource managers to know the range of critical ecological processes and conditions that have characterized particular ecosystems over specified time periods and under varying degrees of human influences. As applied to the management of forested ecosystems in the western U.S., an ecosystem management paradigm emphasizes knowledge of the range of ecosystem conditions prior to significant changes brought on by intensive Euro-American (people primarily of European ethnicity but also of Hispanic, Asian, and African origins) settlement and how these conditions have continued to change during the 20th century (Kaufmann et al. 1994, Morgan et al. 1994, Landres et al. 1999, Swetnam et al. 1999). The timing of major impact of Euro-American settlement varies in the West from the middle 18th to early 20th centuries, but generally begins in the latter half of the 19th century for most areas. Understanding the interactions of humans with natural variaion in determining the current and future conditions of ecosystems is a primary goal of research that supports ecosystem management.

The aim of this report is to assess current knowledge of the historic range of variability (HRV) of forested ecosystems in the Grand Mesa National Forest (GM; Figure 1.1). Our focus is to understand how natural processes have determined the composition, structure, and function of forested ecosystems in the Rocky Mountains of western Colorado over the past several centuries, how Native Americans have influenced these patterns and processes, and how these processes and patterns may have been altered by humans during the 19th and 20th centuries. This report is organized as follows: Chapter 1 is an overview of the goals and conceptual basis for this assessment. Chapter 2 discusses methodology and sources of information used for determining the HRV in this region and some limitations of different types of historical data. Chapter 3 describes the general setting of the GM, including the geology, climate, vegetation patterns, and a brief summary of human settlement and land use. Chapter 4 provides a background on climatic variation over time scales of years to a few centuries for understanding this important potential influence on disturbance and ecosystem parameters. Chapter 5
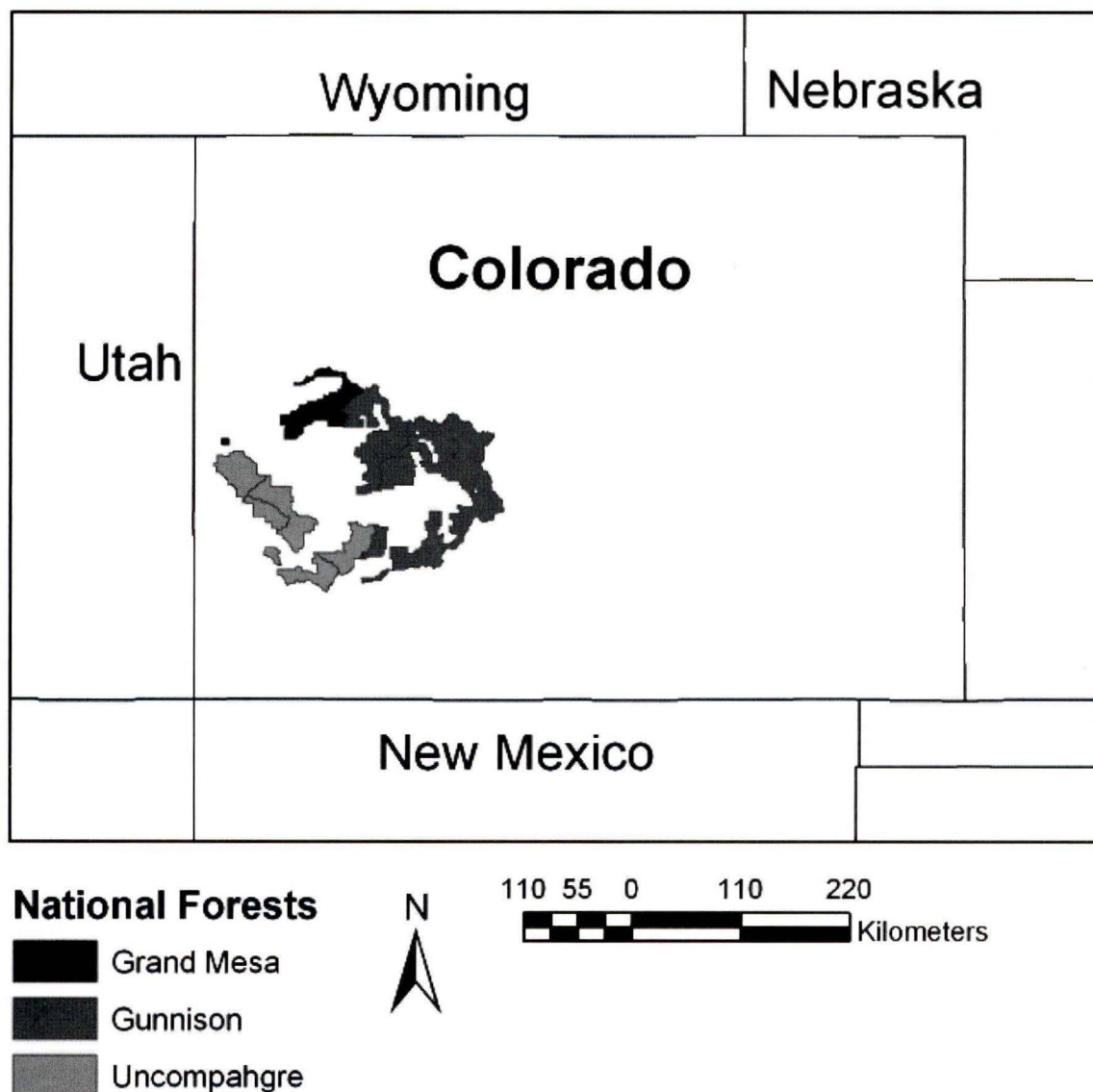
Figure 1.1—Location of Grand Mesa, Gunnison, and Uncompahgre National Forests.

reviews the literature on disturbance patterns and processes for the major forest ecosystems represented in the GM. Chapter 6 discusses what is known about changes in stand, landscape, and regional scale vegetation patterns over recent centuries as driven by both natural and human agents. Chapter 7 summarizes major findings of this assessment in relation to their potential management implications, and also identifies research needs.

Whenever possible, the assessment is based upon data collected in the GM. However, for many topics literature is available only for areas outside the GM or not at all. Furthermore, even for studies conducted within the GM there are limitations to the application of information collected at specific spatial and temporal scales. For example, case studies may reconstruct patterns at a stand or site level (concentrating on an area of 100s m^2) but may not be applicable at a landscape scale (an area of 1000s of hectares). Finally, the amount of detail that is available and that can be incorporated into this report is limited. We expect that this report will serve as a general

synthesis of HRV of forest vegetation for the GM and as an *entrée* into the literature for more detail on specific subjects.

1.1. Concepts of Ecosystem Management and Historic Range of Variability

Modern ecosystem management is based on the recognition that ecosystems are not static and that change occurs due to both humans and natural influences (Swanson et al. 1993, Morgan et al. 1994). Holling and Meffe (1996) argue that past natural resource management practices often have resulted in a loss in the natural variability in ecosystem processes and components. This, in turn, has led to reductions in ecosystem resilience, or the ability of an ecosystem to persist in response to shifts in driving factors or system processes. Holling and Meffe (1996) recognize two definitions for resilience in the ecological literature, and differences in these definitions have the potential to profoundly influence the conceptual basis for natural resource management actions. The first and more traditional definition for resilience they term *equilibrium resilience*, which is the ability for an ecosystem to return to some steady-state equilibrium condition after a disturbance. This concept of resilience is based on constancy and predictability, attributes that they argue are at the core of a "pathology" of natural resource management. In their view, this pathological approach to resource management seeks to reduce variability in ecosystems to make them more predictable and thus more reliable for societal and economic needs. Holling and Meffe further argue that this pathology often has reduced the resilience of ecosystems and led to unforeseen and usually undesirable ecological surprises and crises.

The second definition for resilience, *ecosystem resilience*, refers to the amount of instability an ecosystem can absorb before it changes to a new regime of relatively stable behavior (e.g. a change from forest to grassland; Holling and Meffe 1996). This definition differs from the more traditional concept of resilience in that it emphasizes system dynamics that are inherently unpredictable and may only become apparent for larger systems over longer time periods. The conceptual focus here for natural resource management is to identify actions that adversely impact ecosystem structure or function through changes in the variables and processes that control

ecosystem behavior. As long as a range of variability in system behavior is retained, Holling and Meffe argue that ecosystem resiliency is maximized and ecological surprises or crises can be minimized. The concept of ecosystem resilience recognizes that ecosystems are dynamic and is thus more consistent with modern ecological understanding.

The recognition that ecosystems are dynamic and that change and instability are inherent parts of ecosystem function has been a major shift in perception over the past several decades among scientists and resource managers (Pickett and White 1985, Botkin 1990). Formerly, there was a widespread expectation of a "balance of nature" that was reflected in concepts that stressed stability, such as the climax concept or homeostatic self-regulation of ecosystem properties. Today, ecosystem change is regarded as the norm, and periods of relatively rapid *versus* slow change should be expected and accommodated in management practices.

Understanding the causes and effects of natural variability in ecosystem patterns and processes also provides operational flexibility for management actions and protocols (Landres et al. 1999). Incorporating historical ecosystem patterns into management goals provide an initial strategy for dealing with sustainability of diverse and often unknown species requirements. Managing within boundaries of site variability and history is also probably easier and less expensive to achieve than trying to manage outside of constraints imposed by driving factors of the system (Allen and Hoeskstra 1992, Landres et al. 1999). Historical patterns of ecosystem conditions provide what may be the only viable model for how ecosystems have evolved and perpetuated themselves in the absence of significant human impacts.

1.2. Disturbance and Ecosystem Management

Because many ecosystems in western North America were shaped largely by natural disturbances prior to Euro-American settlement, resource managers in these ecosystems often seek to restore and maintain ecosystems by reinstating select characteristics of disturbance regimes (Kaufmann et al. 1994, Mast et al. 1998). The most reasonable templates for this restoration are descriptions of forest structure and composition and of disturbance regimes that existed

prior to significant land-use changes brought on by Euro-American settlement. Disruption of historical patterns of structure and processes occurred primarily because of grazing, agriculture, logging, and alteration of fire regimes, but may also have been affected by climate change. Although studies of historical range of variability often focus on the time of initial large-scale and permanent Euro-American settlement (the late 19th century for the GM area), ecosystem conditions were in flux prior to this time as well, probably due in part to impacts of Native Americans and certainly due to natural processes such as climatic variations or natural dispersal of species. For example, while lightning is a major cause of forest fires, Native Americans also ignited fires in some areas. The precise effect of Native American burning is uncertain and spatially variable. It has been suggested that Native Americans had an ecologically significant influence primarily at lower elevations and in valley bottoms (Barrett and Arno 1982), but otherwise had only a minimal impact on fire regimes (Baker 2002). Widespread intentional burning in higher elevation subalpine forests would have been more difficult because the greater moisture of those forests is not conducive to the spread of fires during most years. In these higher areas, ignitions were probably not limiting to the occurrence of fire due to frequent lightning strikes (Baker 2002). In any case, the conditions that may be described for key dates such as settlement or the establishment of national forest reserves are only snapshots of changing landscapes. While such a reference point is helpful in understanding how ecosystems have evolved and perpetuated, it is still only snapshot of a naturally fluctuating system and must be used cautiously. The selection of an atypical reference period can lead to a misunderstanding of the ecosystem and inappropriate management decisions. Whenever the data permit, this assessment will consider landscape changes over longer time spans of at least several hundred years.

Understanding vegetation dynamics in western North America requires an appreciation of the roles of both natural and anthropogenic disturbances in these ecosystems (Oliver and Larson 1990, Glenn-Lewin et al. 1992). Disturbance has been defined as: "any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment" (White and Picket 1985:7). In forested landscapes, disturbances remove biomass that creates space for new individuals and releases resources to new and surviving ones. Holling (1992) describes disturbances as mesoscale ecosystem processes that "entrain" microscale vegetative processes responsible for regeneration, growth, and mortality of individual plants to create distinct community and landscape architectures. These architectures are the result of differences in temporal and spatial patterns inherent in disturbance regimes. Historical legacies in vegetation patterns, established by spatial and temporal variability in disturbance occurrences, can persist for long periods until new vegetation patterns are restructured by either climate changes or subsequent intense disturbances. An understanding of vegetation change that results from disturbances is of critical importance for assessing the possible ecological impacts of management activities aimed at suppressing disturbances such as fire or insect outbreaks.

A conceptual framework for considering the characteristics and consequences of disturbance is that of the *disturbance regime*, or the combination of spatial and temporal characteristics of disturbances in a particular landscape (Paine and Levin 1981, White and Pickett 1985). The key potential descriptors of a disturbance regime are: 1) spatial distribution; 2) frequency (mean number of events per time period); 3) size of the area disturbed; 4) mean return interval (mean time between events; the inverse of frequency); 5) predictability; 6) rotation period (time required to disturb an area equivalent to the study area once); 7) magnitude or severity; and 8) the synergistic interactions of different kinds of disturbances and their driving factors (e.g., climate, human ignition sources) (White and Pickett 1985). Variations in these parameters are major determinants of landscape heterogeneity. Although there are numerous case studies of vegetation and ecosystem response to disturbance for the southern Wyoming to southern Colorado sector of the southern Rocky Mountain floristic province, relatively little work has been done on disturbance regimes *per se*. Fire has been the most frequently studied disturbance type, but even for fire, available data are insufficient for a comprehensive quantification of fire regimes across a full range of forest and grassland ecosystem types. Inevitably, assessment of historical ranges of variability of disturbances and their ecological consequences must depend on interpretation of scattered and fragmentary evidence of both a quantitative and qualitative nature.

Four key questions that are related to HRV and that

are critical to forest management in the context of an ecosystem management paradigm are (Veblen 2000):

1. How do disturbance regimes vary spatially? In particular, how does the occurrence of disturbances such as fire and insect outbreak vary along environmental gradients (e.g., low *versus* high elevation) and for different forest types?

2. How have humans altered natural disturbance regimes and how has this varied spatially? For example, has fire frequency (i.e., number of fires per time period) or area burned changed since Euro-American settlement and has this influence varied with elevation or forest type?

3. How do disturbance interactions affect vegetation responses as well as the occurrence and spread of subsequent disturbances? For example, how has modern fire exclusion altered the probability of the occurrence or severity of insect outbreaks?

4. How does climatic variability affect disturbance regimes and vegetation response to disturbances? To what extent might some of the forest health problems usually attributed to management practices (e.g., fire exclusion) be attributed to or exacerbated by climatic variability?

The answers to these questions are very incomplete for the GM and, indeed, for the Rocky Mountains in general. This realization is important both as a caveat to incorporating the tentative knowledge of disturbance patterns into discussions of ecosystem management and as a guide to future research needs.

2. Methodology

A variety of sources are synthesized in this HRV assessment. When possible, we emphasize published sources of information that have been subjected to critical examination in the peer-review process. Information was obtained from scientific and other journals, books, unpublished government reports, and unpublished data sets from the Forest Service or from individual researchers. Some types of information will not have been subjected to peer review, but we attempt to place these data into the context of this assessment using our own evaluations. When there is a choice we cite data sources that are available to the public (e.g., publications, dissertations, and theses) rather than individually held data sets (e.g., works in progress).

Because much of the available data, especially tree-ring data, are from forested areas, HRV is assessed primarily for those ecosystems. In most cases, comparable data for grasslands, meadows, and bodies of water are not available. This lack of data is a major obstacle for assessing HRV for non-forest cover types. Although non-forest vegetation covers a relatively minor part of the GM NF, such habitats undoubtedly account for a large percentage of plant and animal diversity, and thus are of particular importance. Until research provides better insight into the HRV of non-forest cover types, some of the general patterns can be extrapolated to these non-forest vegetation types based on their locations near areas of reconstructed variability. Our interpretations of data and the literature incorporate our personal field experiences in the GM and nearby areas and also benefit from feedback from Forest Service personnel and other people with firsthand knowledge of the GM.

We refer to several scales of analysis in this report. Stand level refers to historical patterns that occur over spatial scales of 0.01 to a few km^2. Landscape scale refers to spatial scales of a few to c. 100 km^2. Regional patterns are those that occur over areas much greater than 100 km^2, and may include historical patterns that are known or inferred to have occurred over much of central and southern Colorado. Knowledge of past patterns is highly uneven for different spatial scales. For example, historical patterns may be well known at a stand scale (e.g., when inferences are made from tree age-structures) but less certain at landscape or regional scales.

Information on historical variability in ecosystem conditions comes from a variety of both quantitative and qualitative sources. Each of these sources has its own limitations that need careful interpretation before use in guiding management decisions. This assessment relies on direct sources of information (e.g., eyewitness descriptions of past events and landscapes) as well as proxy data on the historical range of variability. In the absence of direct measurements of ecosystem variability, proxy data are used to reconstruct historical ecosystem variability from indirect evidence. Well known examples of proxy data include the use of fossil

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pollen to reconstruct past vegetation (e.g., Fall 1997) and the use of tree-ring widths to describe pre-historical climatic variation (e.g., Woodhouse 1993). Although proxy data are quantitative, they are censored in the sense that the full range of data may not be available because we can only sample what has survived to the present. Proxy records are filtered by past environmental processes and loss of evidence through time, often resulting in missing, patchy, or altered records. The accuracy of any environmental reconstruction from proxy records depends on how well this filtering process is understood and modeled (Swetnam et al. 1999).

Information sources utilized in this HRV assessment include: 1) historical records and studies; 2) macro- and micro-fossil records; 3) tree rings; and 4) modern conditions of ecosystem parameters. The limitations and potential biases of each of these sources has been discussed in detail in methodological reviews (e.g., Fritts 1976, Vale 1982, Rogers et al. 1984, Prentice 1988, Fritts and Swetnam 1989, Johnson and Gutsell 1994, Swetnam et al. 1999, Baker and Ehle 2001; Veblen 2003). Below, we briefly comment on the key sources specifically used in this assessment.

2.1. Early Historical Records, Photographs, and Reports

The earliest survey of vegetation conditions of the area of present-day GM is Sudworth's (1900) report on Battlement Mesa Forest Reserve conducted for the U.S. Geologic Survey (Figure 2.1). Although this is the most valuable early description of Grand Mesa National Forest, Sudworth did not use any sampling procedures that yielded quantitative descriptions of past landscape conditions. Nevertheless, his narrative and numerous photographs indicate that he conducted fieldwork over a large part of the Reserve. He also mapped the various cover types of GM and areas of burned and unburned forest. Despite the crude methods available for mapping at the time, Sudworth produced a 1:253,000 scale map which has been shown to have a high degree of reliability in terms of cover type accuracy and positional accuracy (the root mean square error of 30 control points was 173 m; Kulakowski et al. 2004). We digitized, georeferenced, georectified, and rubbersheeted the historical map in a Geographic Information System. The georeferencing procedure, which defines how raster or image data are

situated in map coordinates, was based on aligning 30 control points (e.g. mountain peaks and other reference locations) that were identified on the historical map and on modern topographic maps. The average positional error (root mean square) of the 30 control points was 173 m, which is highly accurate for a historical map of this scale. To additionally assess the accuracy of the historical map, we visited 25 random points in the field to describe their tree species composition. All of these points were correctly classified on the historical map. Sudworth's report contains over 30 photographs, in addition to numerous unpublished photographs, many of which show large expanses of the landscape. Although the historical photographs cannot be used to quantitatively assess past landscape conditions, the abundance of the photos and their scattered locations imply that they represent widespread landscape conditions. We have cautiously interpreted Sudworth's findings in the context of other sources of information such as independent descriptions of the landscape, fire-scar dates, and age structures of modern forests.

2.2. Macro- and Micro-Fossil Records

Few macro- and micro-fossil records of plant and insect remains are available for sites in or near the GM in the southern Rocky Mountains. These studies yield quantitative evidence of past environmental conditions; however, they are censored samples of past environmental conditions that require assumptions in reconstructing vegetation and/or climate. Nevertheless, the broad patterns interpreted from fossil data are useful for describing general patterns of vegetation and climate during the past c. 12,000 years. Although the temporal scales of environments reconstructed from micro- and macro-fossil are often too coarse for understanding specific changes in recent centuries (Kaufmann et al. 1998), we include these studies here to document some of the long-term changes in vegetation patterns that may be continuing during the 20th century.

2.3. Tree-Ring Evidence of Past Environmental Conditions

A large portion of this HRV report relies on tree-ring evidence of past environmental conditions. Tree rings are used to describe the histories of fire, climatic

Extent of Sudworth's 1898 map

■	Extent of map
—	Grand Mesa Forest

N

Collbran ▲

Grand Junction ▲

Whitewater ▲

Cedaredge ▲

20   0   20 Kilometers
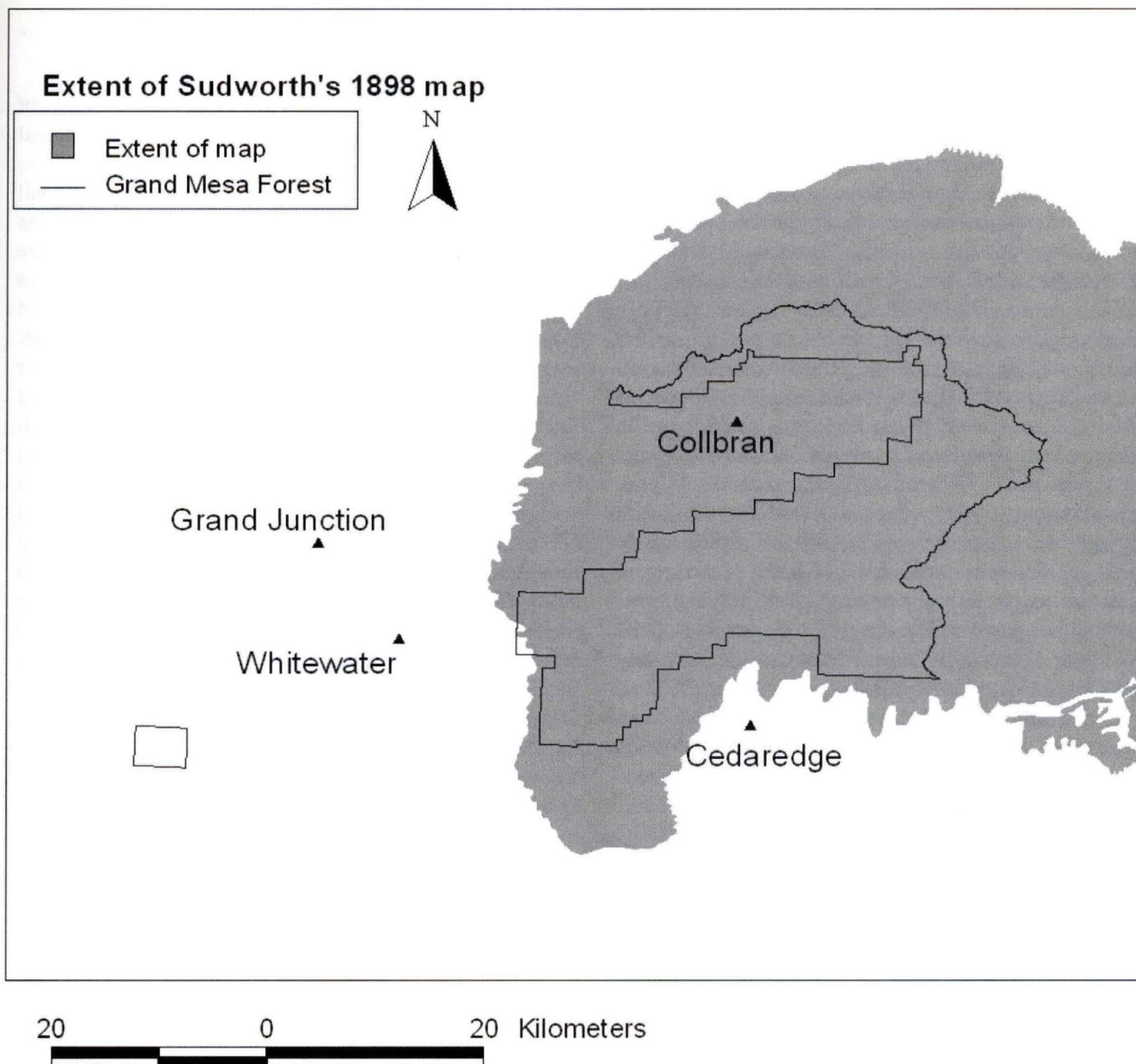
Figure 2.1—Extent of Grand Mesa National Forest (outline) and the extent of vegetation and fires mapped by Sudworth (1900) (gray).

variation, and outbreaks of some insects over the past several hundred years as well as in the evaluation of stand development patterns as influenced by human activities over the past c. 150 years. For assessing climatic variation during the pre-instrumental time period (prior to c. 1890s in the GM), tree rings can provide annually resolved information on climate (e.g. Graybill 1989, Woodhouse 1993). Tree rings document primarily year-to-year variation in moisture

availability and its potential impact on fire occurrence (e.g. Cook et al. 1998, Veblen et al. 2000).

Tree rings also are used to date past fires by dating fire scars and ages of post-fire cohorts. Fire history in forested areas can be described quantitatively on the basis of two types of tree-ring evidence: dates of fire scars (fire-interval approach) or age of stands that presumably regenerated following stand-replacing fires (stand-origin approach). The fire-scar based approach

usually provides annual (or even seasonal) resolution of the dating of past fire events but is limited in its ability to determine the spatial extent of past fires. In contrast, the stand-origin approach utilizes the extent of even-aged post-fire stands to delimit the boundaries of the most recent fire at a site (Johnson and Gutsell 1994) but often does not provide annual resolution of the fire date (Kipfmueller and Baker 1998a). Important limitations of the fire-scar based method include (McBride 1983, Agee 1993, Lertzman et al. 1998, Swetnam et al. 1999, Baker and Ehle 2001): 1) possible elimination of part of the fire record due to logging or extensive tree mortality caused by insect outbreaks or recent intense fires; 2) uncertainty over how representative fire -scar dates are for the larger landscape when scarce old trees are subjectively sampled for fire scars; and 3) incomplete recording of fires due either to fire behavior (e.g. fast moving fires) or lack of fire-scar-susceptible trees at the time of the fire. Thus, fire histories based on fire scars rarely provide a complete and spatially precise record of all past fires within a sample area or a larger landscape. Despite these limitations, a fire history based on an adequate sample size of crossdated fire-scar dates provides a useful quantitative, but filtered, *index* of past trends in fire occurrence.

Fire history studies can document trends and can describe dominant fire severity (high vs. low) but do not necessarily provide precise fire frequency targets for re-incorporating fire at a "natural" frequency. The degree to which such studies can be used to understand an actual fire regime depends on how well the filtering process is understood (Lertzman et al. 1998, Swetnam et al. 1999, Baker and Ehle 2001). Summary statistics of fire history, such as mean fire interval or fire rotation, are inadequate descriptors because the ecological understanding of the effects of past fires requires a much more comprehensive description of a fire regime including spatial pattern, severity, effects on tree demography, and interactions with other disturbances (Veblen 2003). Because of these uncertainties, we make only limited and cautious comparative use of mean fire intervals for characterizing fire occurrence in different time periods or different ecosystems. Instead, we stress the importance of considering trends in regional annual indices of fire occurrence. We also stress the interpretation of fire-scar evidence in conjunction with other lines of evidence such as tree population age structures, changes in growth patterns of individual

trees, and historical landscape photographs, all of which may provide some indication of the ecological impact of past fires.

In contrast to the application of the fire-interval approach to forests dominated by fire-resistant pines and Douglas-fir (*Pseudotsuga menziesii*), fire history studies conducted in dense spruce-fir (*Picea engelmannii-Abies lasiocarpa*) forests in the southern Rocky Mountains are usually derived from dating post-fire stands, sometimes in combination with fire-scar dating (Romme and Knight 1981, Kipfmueller and Baker 1998a). Potential sources of error in the stand-origin approach include: 1) undetermined time lags in tree establishment after stand-replacing fires; 2) erroneous determination of total tree age due to tree core samples that do not intercept the pith right at the root/shoot interface to determine actual germination date; 3) erroneous ring counts due to missing or false rings if samples are not crossdated; 4) possible confusion of a post-fire cohort with tree establishment following other types of disturbance or climatic variation; and 5) destruction of evidence of earlier fires by more recent burning (Veblen 1992, Goldblum and Veblen 1992, Kipfmueller and Baker 1998a). Due to these and other limitations (Huggard and Arsenault 1999) we cautiously interpret estimates of fire frequency and fire rotation on the basis of the stand-origin approach.

2.4. Inference from Modern Conditions of Ecosystem Parameters

This HRV assessment makes use of modern (i.e., post 1950) descriptions of ecosystems in the GM such as data on current forest structure and composition. The fundamental limitation of these data sources is the lack of any comparable pre-20th century data bases for comparison. However, tree growth and age data from living as well as dead trees collected during the modern period can be used to reconstruct past forest structures (e.g. Kulakowski and Veblen in press). Such studies can provide valuable insights into past forest composition and structure as well as disturbance histories. The principal limitations of intensive stand reconstructions include: 1) limited spatial extent; 2) incomplete preservation of evidence with increasing time before present; and 3) at some sites, incomplete or no information on potentially important past influences of livestock on stand structure. Decay and disappearance of dead trees results in approximate and

partial reconstructions of past forest conditions. Consequently, the quantitative limitation to their precision needs to be considered when interpreting their implications for ecological restoration. The most serious limitation of such an approach for GM is simply the lack of abundant tree-ring data collected to reconstruct past stand conditions and local disturbance histories. At a broad spatial scale, U.S.D.A. Forest Service's RMSTAND, tree age data provide the potential for interpreting general features of past forest structure. The RMSTAND data set includes the approximate ages of forest stands based on increment core samples. However, these data have severe limitations for understanding forest patterns and dynamics. For example, 1) cores collected at breast height (as in the RMSTAND database) may miss many decades of tree age, especially for slow-growing trees in the subalpine zone (Veblen 1986a); 2) the small number of trees sampled in each stand do not constitute an objective sample, and may not include the oldest tree in the stand (Goldblum and Veblen 1992, Kipfmueller and Baker 1998a); and 3) core samples that did not intercept the pith do not accurately date the tree's age at coring height (Veblen 1992). Furthermore, our personal experiences indicate that ring counts done in the field are often highly inaccurate. Given these problems, we make only limited and cautious use of tree age data from the RMSTAND database.

Due to the limitations of all sources of information on HRV, we do not rely on only a single line of evidence to reconstruct historical variability. Instead, we use inferences and evidence from multiple sources to draw conclusions about general to specific patterns that resulted from past processes. Where specific information on past forest conditions is lacking, we sometimes draw inferences from known past processes. Some interpretations are presented as hypotheses, and new data and studies may contradict or support interpretations presented here.

Habitat type classification is sometimes used in the western US to describe plant communities. Habitat typing may be useful for comparing GM ecosystems to other areas where a variety of types of studies may exist (e.g. fire history, successional studies). Such a comparison may help to assess the applicability of those studies to GM. However, this system cannot be relied upon extensively because its premises are at odds with current understanding of succession, disturbance, and interactions between overstory and understory vegetation (Cook 1996).

Finally, a critical issue in assessing the range of historic variability in the GM is the degree to which conclusions based on studies conducted elsewhere can be applied to the GM. For example, there is more abundant literature on fire history in spruce-fir ecosystems in Routt and White River National Forests (Veblen et al. 2001, Kulakowski et al. 2003, Kulakowski and Veblen 2002) and in the Medicine Bow National Forest (Romme and Knight 1981, Kipfmueller and Baker 2000). However, differences in climate, vegetation structure, and human settlement mean that the conclusions from these studies cannot be uncritically accepted for the GM. Consequently, in this report, whenever possible, we rely on studies conducted in the GM or nearby areas. For topics that lack on-site studies, we necessarily must refer to studies conducted elsewhere. We will make explicit what we consider to be general patterns for ecosystems similar to those found on the GM *versus* patterns that have been documented specifically in the GM. We consider findings from elsewhere as hypotheses that need to be critically examined for the GM. These hypotheses can be evaluated according to similarity of sites in terms of ecosystem attributes (e.g., habitat types), human history, and regional climates. Even for studies conducted within GM, there are important limitations to applicability of the data over the entire GM. Throughout this report we compare present ecosystem conditions with those of the probable range of historical variation and categorize those conditions as being "inside" or "outside" of HRV, or trending towards one of those two conditions.

3. Environmental and Cultural Setting of the Area

3.1. Geology and Soils

The GM is located in western Colorado east of Grand Junction, northwest of Gunnison NF, and is entirely confined to the western side of the Continental Divide (Figure 1.1). The GM includes an area of about 160,000 ha (395,200 acres) and is administratively combined with the Uncompahgre and Gunnison National Forests. In elevation this Forest ranges from the woodlands near the Colorado and Gunnison River valleys at around 2,100 m (7,000 feet) to subalpine environments on the Grand and Battlement Mesas and

peaks such as Leon Peak at over 3,400 m (11,000 feet) (Figure 3.1).

The dominant physiographic feature of the GM is the high plateau formed by the Grand and Battlement Mesas. The large basaltic plateaus of Grand and Battlement Mesas were formed by uplift and associated lava flows during the Tertiary Period (3-70 million years ago) (Chronic and Chronic 1972). These plateaus are underlain by Tertiary sandstone and shale of the Wasatch and Green River Formations (Chronic
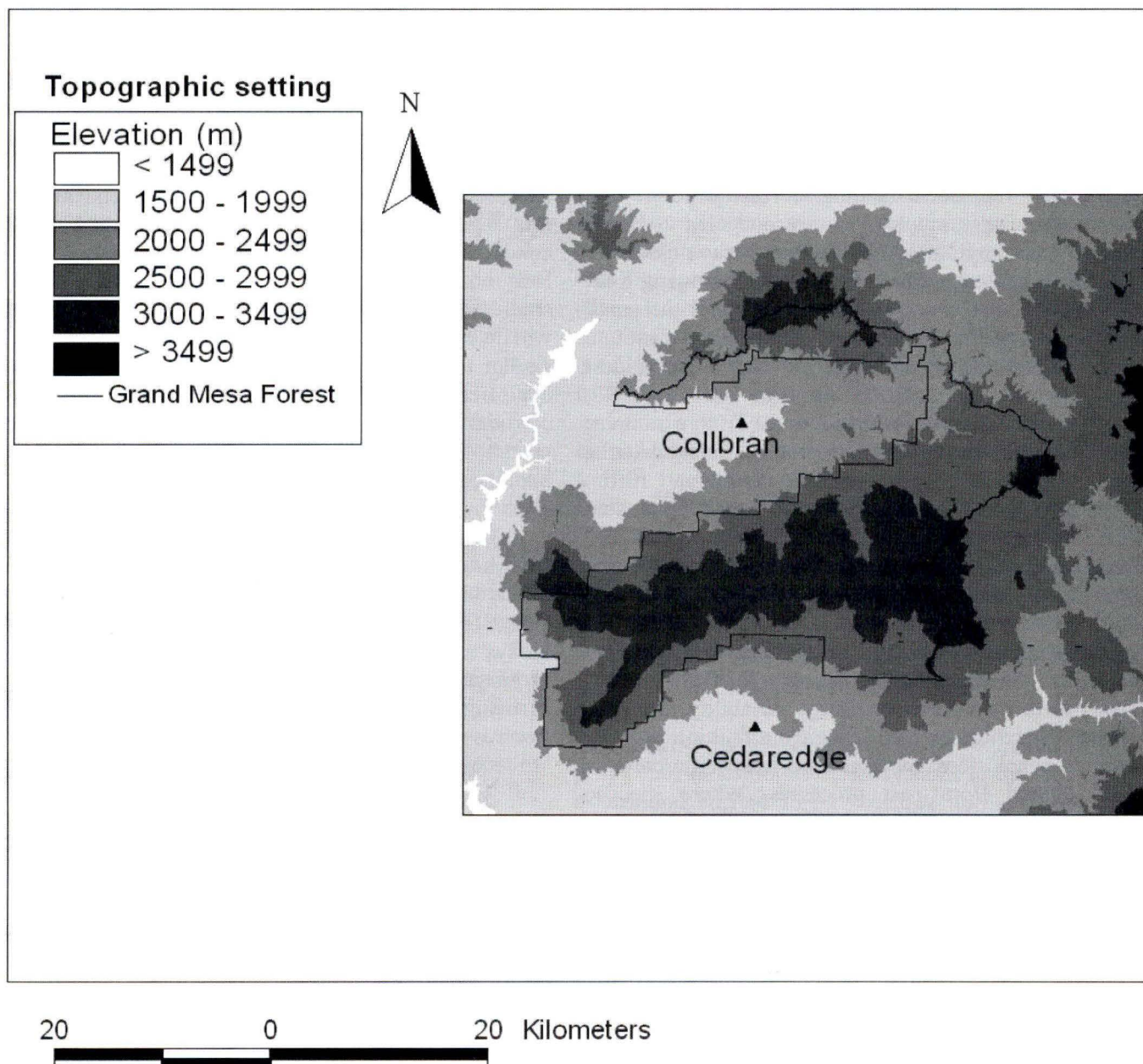

Figure 3.1—Topography of Grand Mesa National Forest and adjacent areas.

1980). These, in turn, rest on Cretaceous Mesaverde Group and on Cretaceous Mancos Shale, which continues to the fertile valleys below. The valleys around the mesa were subsequently carved by the Colorado and Gunnison Rivers flowing over the soft shale. Soils of the GM are dominantly Gateway-Cebone-Bangston association, which are moderately deep and well-drained soils (Spears and Kleven 1978). This soil association is a mix of the alfisol Gateway loam, and the mollisol Cebone loam and Bangston sandy loam. Generally, the mollisol is associated with grasslands and aspen (*Populus tremuloides*) while the alfisol is associated with other woody vegetation types.

The GM includes important watersheds that collect water for urban and rural areas including Grand Junction, Fruita, Pallisade, and other areas of western Colorado. Many of the 300 lakes and reservoirs on the GM are diverted or dammed for irrigation and urban use.

3.2. Climate

The high elevation and continental interior location of the GM predominantly control its climate. Its interior location results in relatively dry conditions and wide differences between summer and winter temperatures. Synoptic-scale climate is dominated in the winter by westerly flow from the Pacific. During winter when northern Pacific maritime air masses meet the Rocky Mountains, orographic precipitation occurs on the western slope. In the summer, with the development of the North American monsoon, convection cells create locally heavy thunderstorms often with brief but intense precipitation (Barry et al. 1981). The regional climate in Colorado is also influenced by mid-tropospheric pressure anomalies associated with the Pacific-North American pattern (PNA; Cayan 1996).

There are strong seasonal differences in temperature and precipitation in the GM. Mean monthly temperature records from climate stations in or near the GM (Table 3.1) show similar seasonal trends (Figure 3.2). Precipitation generally increases from its minimum during the winter months of December and January to peaks in spring and late summer (Figure 3.3). The July-August peak in precipitation is associated with convective storms under the influence of the North American monsoon.

Seasonal moisture availability is greatly influenced by elevation. Because high-elevation sites have cooler

late-spring and summer temperatures and higher amounts of winter precipitation, these sites would also be expected to have higher moisture availabilities during the early part of the growing season. Additionally, the lag in snowmelt at higher elevations would be expected to supply soil moisture in mid-spring (i.e., June), when precipitation is greatly reduced. Low elevation areas experience a water deficit in June if snowpack has disappeared and precipitation is absent. Thus, tree growth is more sensitive to June moisture deficits at low than at higher elevations on Colorado's east slope in the Front Range (Donnegan 2000). Analogously, variation in spring (April-June) moisture availability has a greater impact on fuel conditions at low elevations (below c. 2500 m) whereas mid-summer (July-August) has a relatively greater impact on fuel conditions at high elevations on the east slope (above c. 3000 m; Veblen et al. 2000, Sherriff 2000). While such studies have not been conducted on the GM, this general relationship of elevation, snowpack, moisture availability is likely to be similar.

3.3. Vegetation

3.3.1. Spatial Patterns

The GM encompasses an array of vegetation types that include pinyon-juniper (*Pinus edulis-Juniperus* spp.) woodlands, Gambel oak (*Quercus gambelii*) shrubland, aspen (*Populus tremuloides*) forests, Engelmann spruce-subalpine fir (*Picea engelmannii-Abies lasiocarpa*) forests, and grasslands (Table 3.2; Figure 3.4). Each of these major vegetation types can be sub-divided into numerous habitat types (e.g. Komarkova et al. 1988) that occur in association with various abiotic characteristics and are indicated by variation in the non-arboreal plant species composition. While variations in vegetation patterns and successional development certainly exist within each forest type, we limit our discussion to the general successional patterns for each of the major forest types. In the GM, vegetation pattern is strongly controlled by elevation and moisture gradients. These factors play a major role in the amount, distribution, and persistence of moisture available to vegetation. Elevation also affects temperature that controls growing season length and the amount of freezing that species must be able to tolerate.

A gradient of forest types can be recognized, which

Table 3.1—Details of the instrumental climate stations used for describing temperature and precipitation trends in the Grand Mesa area. Data from: the Colorado Climate Center of Colorado State University (http://climate.atmos.colostate.edu/).

Station	Elevation (feet)	Period of the Record	
		Temperature	Precipitation
Bonham Reservoir	9820 feet		1963 - 1994
Cedaredge	6180 feet	1898 - 1994	1898 - 1994
Collbran	6140 feet	1900 - 1999	1893 - 1999
Grand Junction WS	4850 feet	1899 - 2003	1899 - 2003
Palisade	4720 feet	1929 - 2003	1929 - 2003

Figure 3.2—Mean monthly temperature for three climate stations in or around Grand Mesa National Forest. Record lengths are 1900-1999 for Collbran, 1899-1999 for Grand Junction WS, and 1898-1994 for Cedaredge.
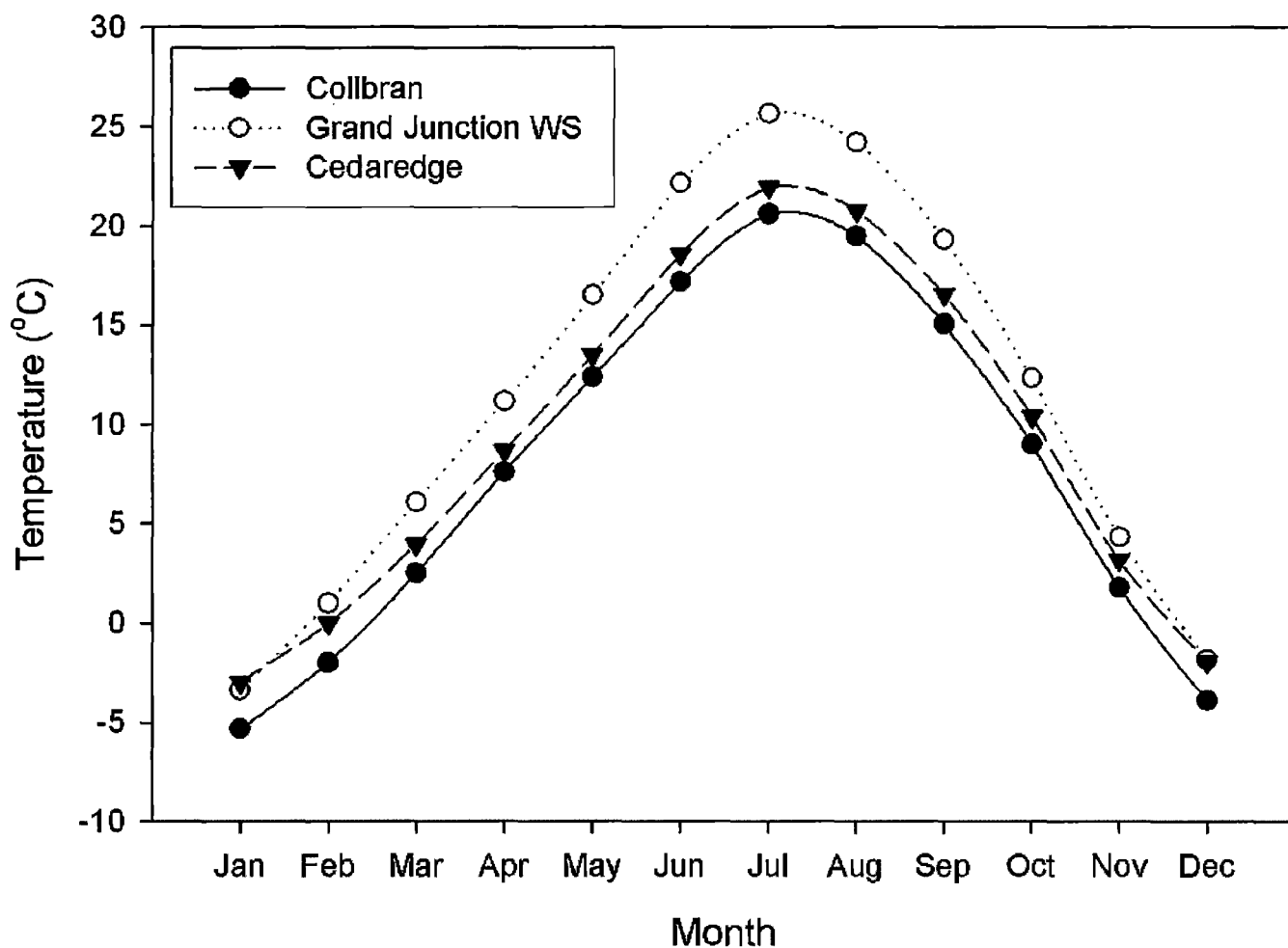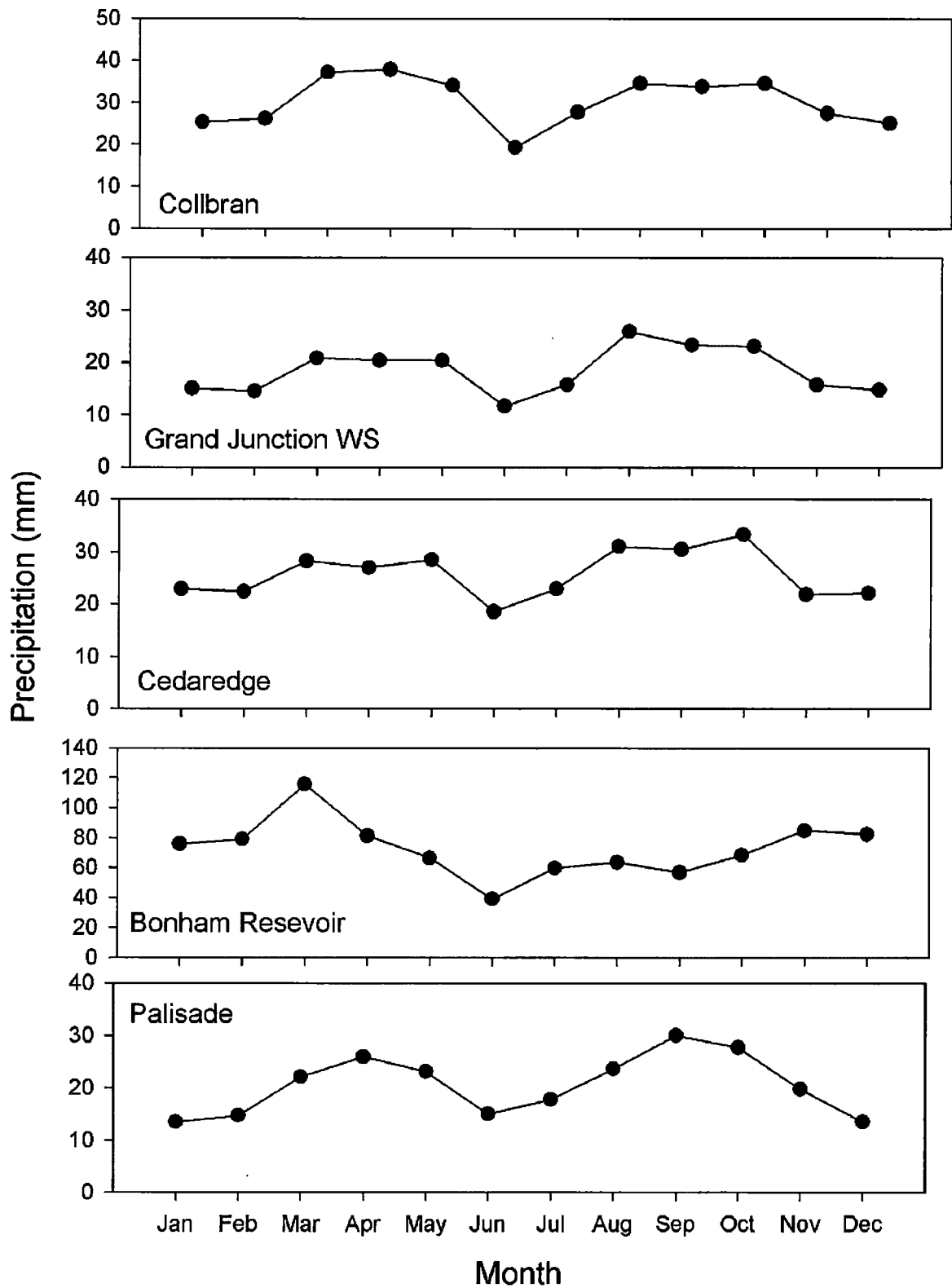
Figure 3.3—Mean monthly precipitation for three climate stations in or near Grand Mesa National Forest. Record lengths are 1893-1999 for Collbran, 1899-1999 for Grand Junction WS, 1898-1994 for Cedaredge, 1963-1994 for Bonham Resevoir, and 1929-1999 for Palisade.

Table 3.2—Cover types of the Grand Mesa National Forest (from USDA Forest Service RIS database). Percentages refer to the sum of the cover types in the table and exclude land surfaces of water, rocks, and other non vegetated areas.

Forest Type	Total Area (ha)	Percent
Aspen (*Populus tremuloides*)	48,642	30.6
Blue spruce (Picea pungens)	22	0.0
Cottonwood (*Populus angustifolia*)	171	0.1
Douglas-fir (*Pseudotsuga menziesii*)	1,743	1.1
Forblands	5,733	3.6
Grasslands (*Gramineae*)	13,739	8.6
Oak (*Quercus gambelii*)	24,403	15.4
Pinyon-Juniper (*Pinus – Juniperus*)	14,207	8.9
Ponderosa pine (*Pinus ponderosa*)	14	0.0
Sagebrush (*Seriphidium tridentatum*)	1,421	0.9
Shrublands	6,126	3.9
Spruce – Fir (*Picea engelmannii - Abies lasiocarpa*)	41,003	25.8
Willows (Salix spp.)	1,741	1.1
Total	158,964	100.0

Pinyon - Juniper Woodlands

■ Pinyon - Juniper
— Grand Mesa Forest

N

Collbran

Grand Junction

Whitewater

Cedaredge
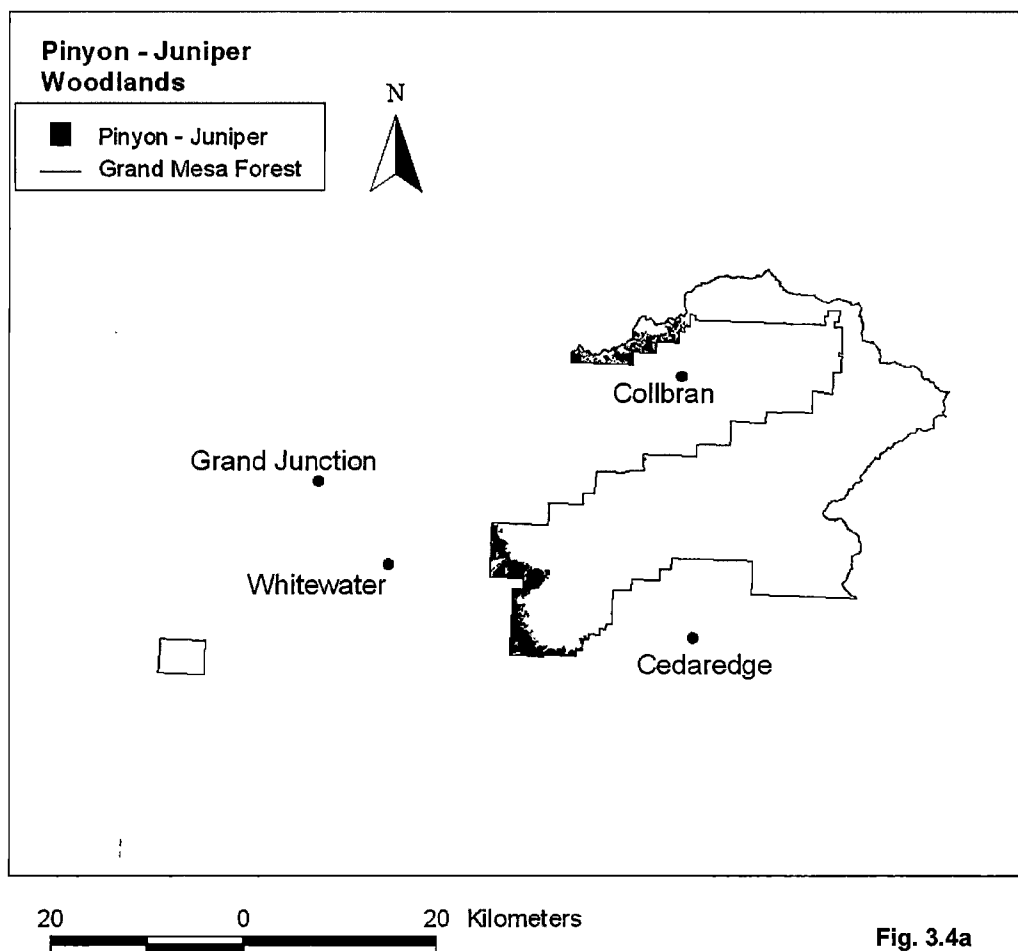
20   0   20 Kilometers

Figure 3.4—Distribution maps of the major vegetation cover types of GM based on the RIS database:
a) pinyon-juniper woodlands; b) oak woodlands;
c) aspen forests;
d) Engelmann spruce-subalpine fir forests;
and e) grasslands.

Fig. 3.4a

14

Oak Forests

- ■ Oak
- —— Grand Mesa Forest

N

Collbran

Grand Junction

Whitewater

Cedaredge

Fig. 3.4b

20   0   20   40 Kilometers
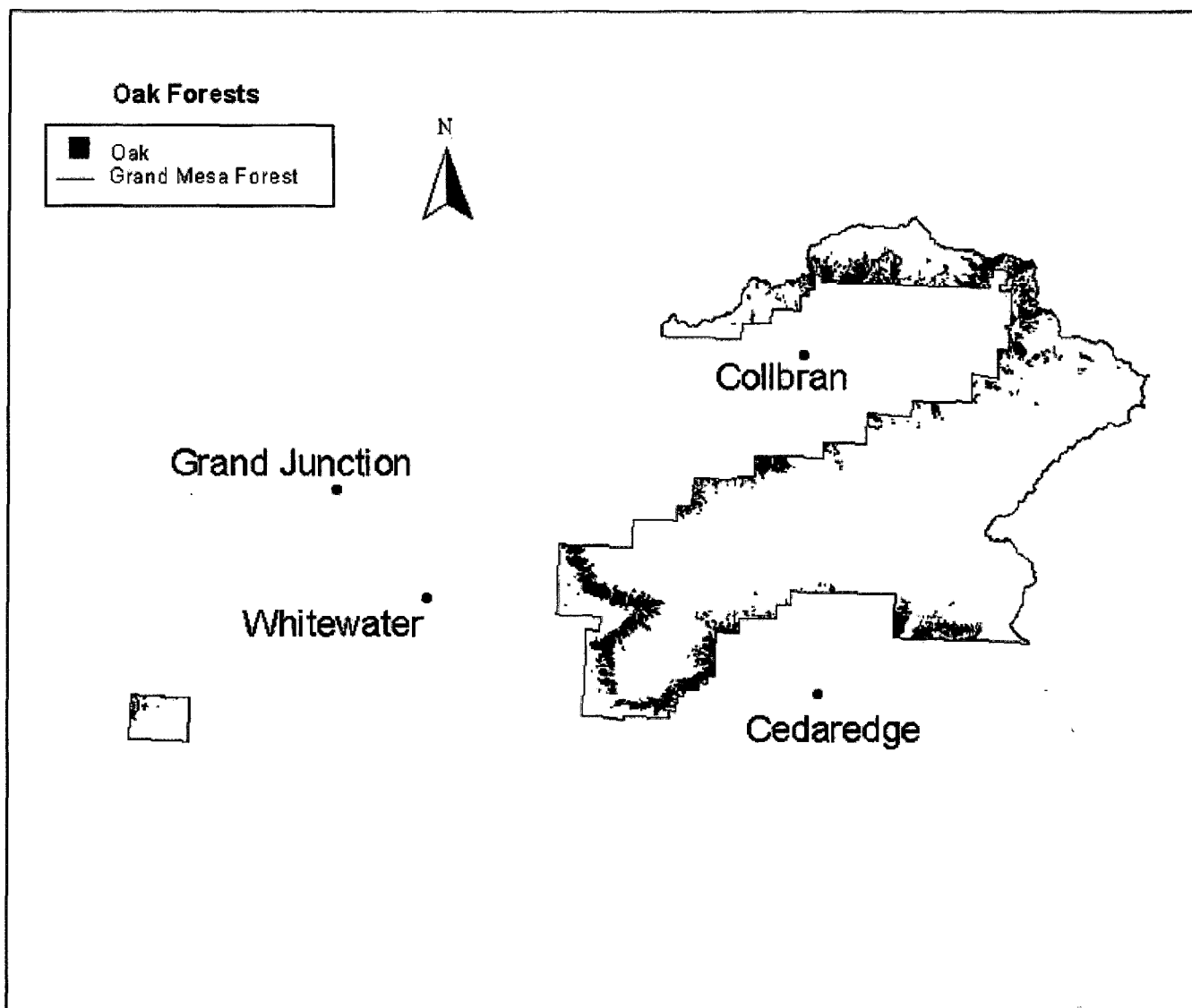
is strongly related to elevation. The montane zone (also known as pygmy zone or pygmy woodland zone) is dominated by relatively open woodlands of pinyon pine and juniper that generally occur below 2500 m. Typically, the relative dominance of pinyon within this zone increases as elevation increases. At higher elevations, this zone grades into communities dominated by Gambel oak, which forms relatively pure stands of open woodlands to denser shrubland ranging in elevation from approximately 2300 - 2800 m. Upper elevation subalpine forests are dominated by quaking aspen, Engelmann spruce, and subalpine fir. Quaking aspen occurs mainly between 2500 - 3100 m. It forms pure stands in its lower elevational distribution and forms mixed stands with Engelmann spruce and subalpine fir at higher elevations. Aspen can grow at xeric sites where it occurs as relatively small trees, and it also occurs on more mesic sites, forming dense stands of tall trees. As discussed below, in its upper elevational range, aspen can be seral to spruce and fir. The forests at highest elevation (above 2800 m) in the GM are primarily co-dominated stands of Engelmann

15

Aspen Forests

Aspen
Grand Mesa Forest

N

Collbran

Grand Junction

Whitewater

Cedaredge
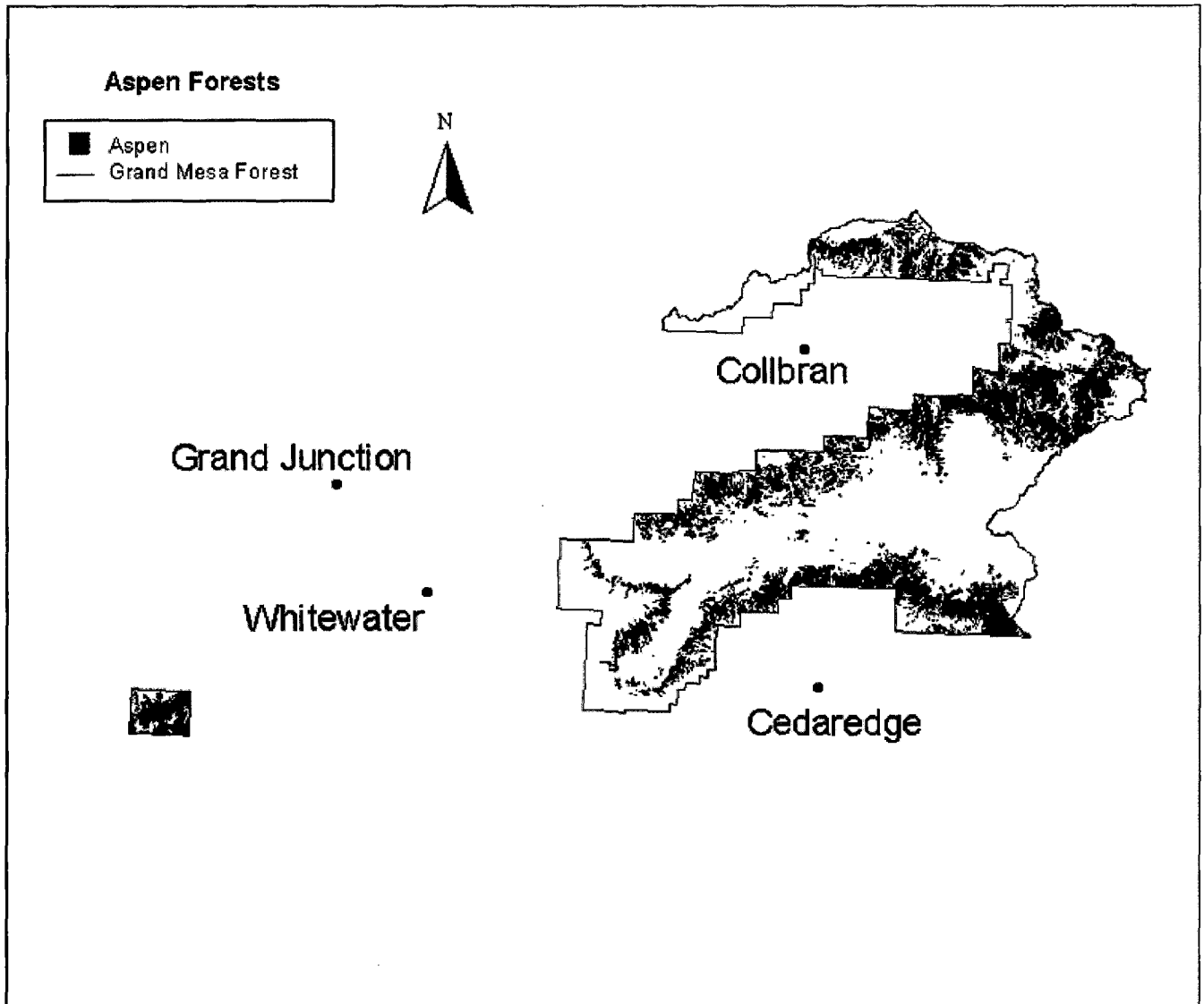
Fig. 3.4c

20   0   20   40 Kilometers

spruce and subalpine fir. These stands are often relatively dense and in some areas occur in a mosaic of grasslands. Grasslands occupy *c.* 9% of the GM and occur primarily on top of the Mesa, interspersed with spruce-fir forests. The most extensive forest cover types of the GM are aspen and spruce-fir. Additionally, small areas of Douglas fir (*Pseudotsuga menziesii*) occupy rocky sites and narrow bands of narrow leaf cottonwood (*Populus angustifolia*) occupy some riparian habitats.

3.3.2. Stand Development and Successional Patterns of the Major Forest Types

In general, plant succession is currently recognized as a highly contingent process that varies depending on site and historical conditions (Pickett et al. 1987, Glenn-Lewin et al. 1992, Cook 1996, McIntosh 1999). Successional pathways vary temporally depending on "initial conditions", such as site heterogeneity and landscape conditions related to the nature and severity

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Engelmann Spruce - Subalpine Fir Forests

■ Spruce - Fir
── Grand Mesa Forest

N

Collbran

Grand Junction

Whitewater

Cedaredge

Fig. 3.4d

20   0   20   40 Kilometers
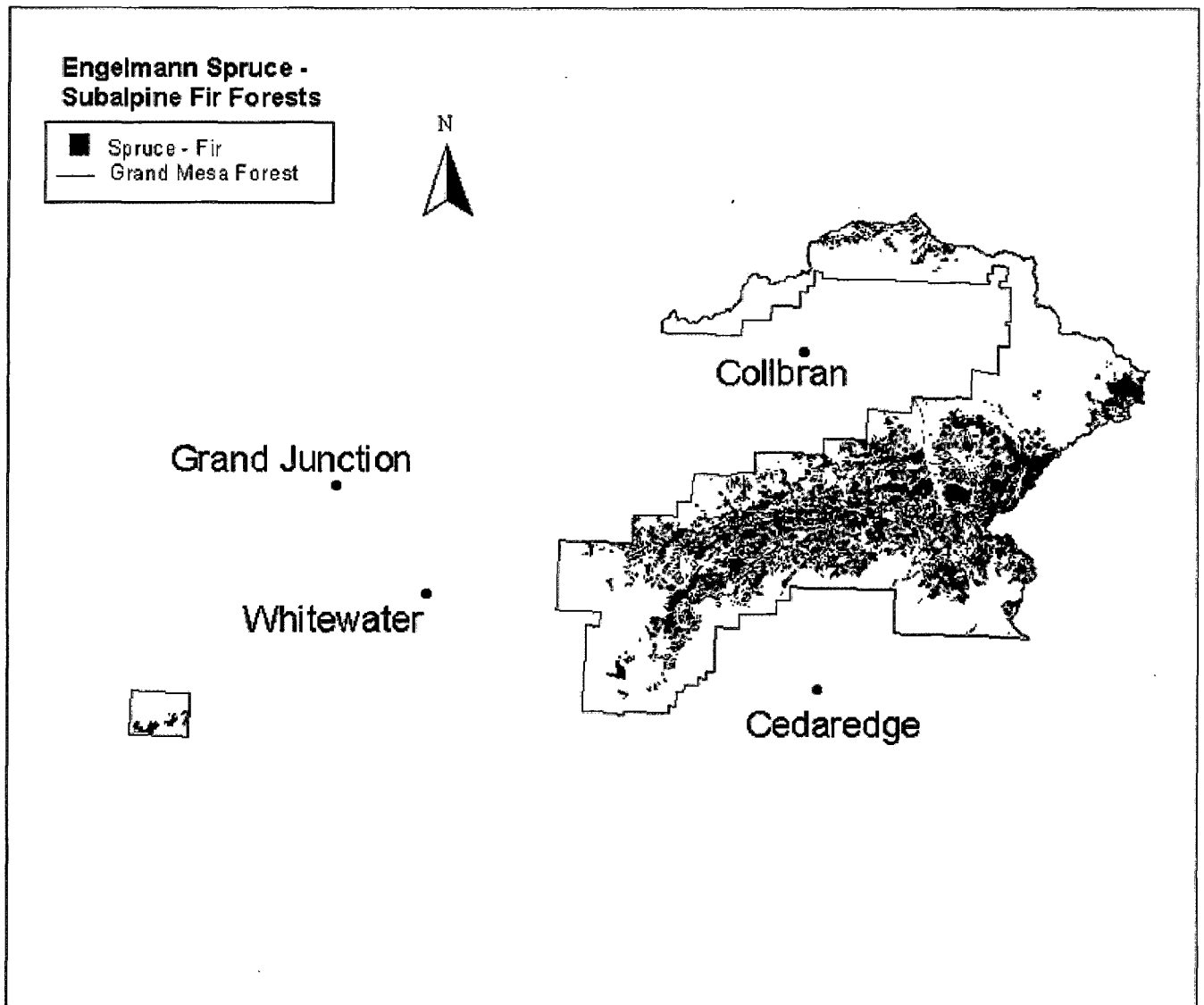
of past disturbances, and long-term climatic variation. Succession also varies with often steep environmental gradients that occur over short distances in the Rocky Mountains. Slight variations in aspect or topographic position have been shown to result in major differences in stand development and successional pathways (Whipple and Dix 1979, Peet 1981, Veblen 1986a, Veblen and Lorenz 1986, Donnegan and Rebertus 1999). Despite these sources of variability, similarities of life history traits for members of the same species in the southern Rocky Mountains often result in relatively predictable successional patterns. An important distinction of the GM in comparison with many other subalpine and upper montane forested zones in Colorado is the lack of lodgepole pine (*Pinus contorta*). There are several reviews of stand development and species replacement in the southern Rocky Mountains (Peet 2000, Veblen and Lorenz 1991, Knight 1994), and in this section we will only briefly summarize some of the major patterns. Our

Grasslands

☐ Grasslands
—— Grand Mesa Forest

N

Collbran

Grand Junction

Whitewater

Cedaredge

Fig. 3.4e

20   0   20   40 Kilometers
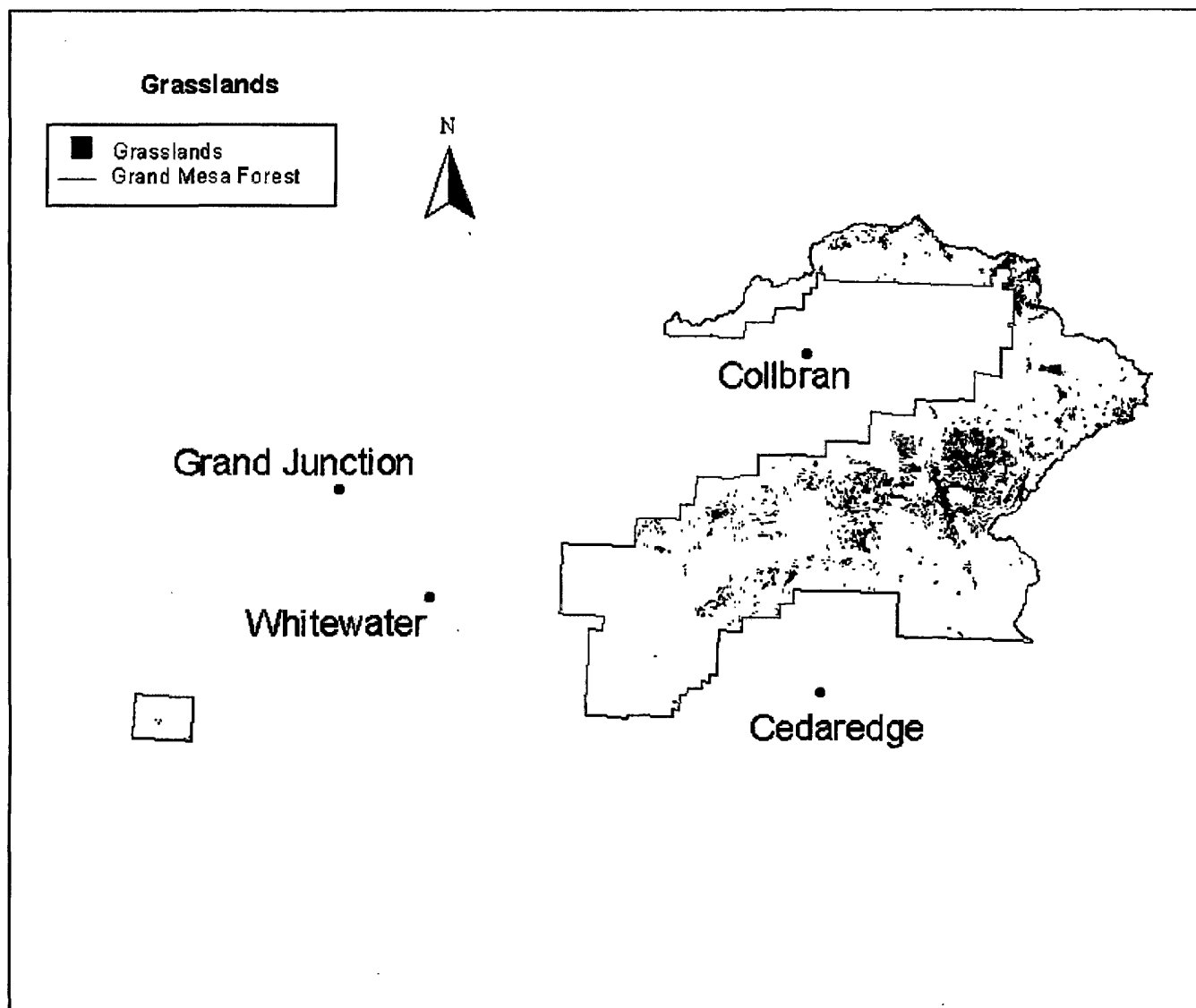
goal is to provide some background on successional patterns for understanding the consequences of natural disturbances as well as of human impacts through logging, fragmentation, and altered fire regimes. The most important natural coarse-scale disturbances triggering succession in the Colorado Rockies are fire, insect outbreaks, windstorms, and, in particular habitats, flooding and snow avalanches.

Pinyon pine-juniper woodlands: Pinyon-juniper woodlands occur over *c.* 19 million ha primarily in seven western states (Evans 1988). A major caveat regarding discussion of pinyon-juniper woodlands is that several different pinyon-juniper woodland types exist. For example, the pinyon-juniper in the Great Basin is composed of a different species of pinyon and juniper than what is found in the Grand Mesa. The pinyon in the Great Basin is a single needle pine that has evolved to withstand much drier conditions due to a moisture regime that is quite different than that of Grand Mesa (See Section 3.2). Thus pinyon-juniper woodlands in different climatic regions are likely to be

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ecologically different and therefore are likely to experience different disturbance regimes. However, due to the dearth of studies within the GM, we must draw on data from other physiographic regions. We do so cautiously, with explicit urging that local research must be conducted on the GM in order to describe the HRV of this National Forest.

Within the GM, the pinyon-juniper cover type is especially widespread in the western part of the Forest (Figure 3.4a). Nearby, on the Uncompahgre Plateau and in the Gunnison Gorge and Black Canyon of Gunnison National Park, age structures suggest that many woodlands originated following infrequent and severe fires, while other woodlands show no evidence of severe fire (Eisenhart 2004). These latter stands can be relatively old (100 - 1000 years) and can form self-replacing stands. Research from Mesa Verde NP shows that in old-growth pinyon-juniper woodlands juniper is longer lived while pinyon experiences higher rates of mortality (Floyd et al. 2003). Thus, age structures of such stands follow a negative exponential distribution with much more abundant young trees than old ones. Increases in stand densities are expected for slow stand development in pinyon-juniper woodlands and do not necessarily reflect suppression of surface fires (Eisenhart 2004). Establishment within pinyon-juniper stands can be episodic and may be related to climatic variability. Climatic variability, at all temporal scales, has been important in influencing the distribution and density of pinyon-juniper woodlands across its range (Betancourt 1987, Betancourt et al. 1993, Tausch 1999). Of several climatic factors, moisture has been suggested to be the most important to pinyon pine's speed of germination and seedling survival (Meagher 1943). A severe drought in the 1950s caused massive die-off of pinyon and juniper throughout the Middle Rio Grande Basin, south of the GM, particularly in the lower distributional limits of pinyon (Betancourt et al. 1993). Wetter conditions in the same area, which began in the 1970s, have been more favorable for pinyon and juniper establishment and growth (Gottfried et al. 1995). A current regional-scale drought is likely to be contributing to a widespread and ongoing mortality of pinyon in the Southwest, including southwestern Colorado (Ogle 2001, Breshears et al. 2005). The ongoing drought is resulting in more severe and more extensive tree mortality than the drought of the 1950s (Breshears et al. 2005). Although in terms of precipitation, the current drought is of a lower magnitude than the

drought of the 1950s, it is accompanied by higher temperatures. The combined effect of reduced precipitation and anomalously high temperatures is the most likely explanation for the high mortality of pinyon pine in this area. Shade has also been suggested to have an important influence on seedling survival (Meagher 1943). Thus, disturbance that leaves coarse debris may hasten re-establishment of trees and may therefore have a significant influence on post-disturbance dynamics (Eisenhart 2004).

Fire was historically the most important disturbance in pinyon-juniper woodlands prior to the introduction of heavy livestock grazing. Fires in pinyon-juniper woodlands can be either severe and stand-replacing in dense stands (Wright and Bailey 1982, Floyd et al. 2000, 2004) or can be less severe or patchy on less productive and open sites (Johnsen 1962). However, a recent review of the literature has shown that there is little or no evidence for spreading low-severity (surface) fires over most of the distribution of pinyon-juniper woodlands in the West (Baker and Shinneman 2004). Based on our field observation in the Grand Mesa, pinyon-juniper woodlands appear mostly characterized by stand-replacing rather than surface fire, although the frequency of these fires may vary spatially with patterns of lighting ignition. To our knowledge, no comprehensive study of pinyon-juniper or Gambel oak woodland fire history has been conducted in GM. Therefore it is not clear whether the fire regime in these woodlands is within the HRV. However, work in other pinyon-juniper woodlands, such as MVNP and the Black Canyon of the Gunnison National Park, suggests that the fire regime of some nearby woodlands may presently be within the HRV, despite a policy of fire suppression in the early 20th century. As further explained in section 5.3, we conclude that the recent (i.e. post-1920) fire regime is not significantly outside of the range of variability of fire over the past few centuries.

Overgrazing by livestock can reduce herbaceous ground cover, therefore altering the fire regime by reducing fuel continuity. The reduction in herbaceous cover can also reduce competition and therefore favor establishment and growth of pinyon and juniper (Gottfried et al. 1995). Although exact numbers are not available for the late 19th century, the density of livestock in the GM area was greater during this time than in other Forest Reserves, such as the White River Forest Reserve (Sudworth 1900). In the 1940s, 30,000 head of cattle and 30,000 sheep were permitted in the

GM. The number of livestock has steadily declined and in 2002, 10,169 head of cattle and no sheep were grazed in GM between June 15 and Oct 15 (USDA Forest Service *unpublished data*).

Gambel oak shrubland: Gambel oak covers an area of *c.* 3.76 million ha (*c.* 9.29 million ac) in the western United States and 21% of this area is found in Colorado (Harper et al. 1985). Over its range, it occurs at elevations of *c.* 1,000 to 3,000 m (3,300 to 9,900 ft), normally in an elevational zone between pinyon-juniper woodlands and aspen forests, as it does in the GM (figure 3.4b). This oak is sensitive to drought (Neilson and Wullstein 1983) and its distribution is probably limited at lower elevations by moisture stress and at higher elevations by competition.

In the northern part of its range and at highest elevations, reproduction from seed is rare. However, in the southern portion of its range (Arizona and New Mexico) reproduction from seed is more common, probably because of more common summer rains (Neilson and Wullstein 1986, Harper et al. 1985). Neilson and Wullstein (1986) therefore suggest that Gambel oak in the northern part of its range may have established in the mid-Holocene when summer rainfall was higher than today. Based on aerial photograph interpretation in Utah, Gambel oak has been reported to have expanded its lower elevational range during the 20th century (Rogers 1982). Repeat photographs of the GM area also suggest that Montane woodlands may possibly be expanding their lower elevational extent (Bradford 1998). However, it is not clear whether this expansion is due to changes in climate, the fire regime, grazing, or a combination of influences.

Reproduction of Gambel oak is usually vegetative, resprouting from rhizomes, especially after fire (Harper et al. 1985). Following fire, the oak sprouts vigorously, grows quickly, and therefore leads to rapid reestablishment of the community within *c.* < 20 years following the fire. In the GM area, annual growth of Gambel oak has been shown to increase following fire and remain above pre-fire levels for at least 10 years following the disturbance (Kufeld 1983). Fire is sometimes used to open tall dense stands of oak shrublands to make them more accessible and to improve forage for elk, deer and cattle (Kufeld 1983). Following intense fire in pinyon-juniper woodlands in Colorado, Gambel oak may initially establish from existing rhizomes and eventually be succeeded by pinyon pine (Floyd et al. 2000).

Gambel oak normally occurs as a tall shrub or short tree (normally < 20 feet tall) in GM. Oak stands are generally dense when young, but the individual trees become relatively slender and erect at *c.* 75-100 years of age, allowing more light to reach the forest floor (Floyd 1982). At this stage in its development, oak may facilitate the establishment of pinyon pine seedlings and may eventually be successional to the pine on certain sites (Floyd 1982).

Generally, little is known about the disturbance history and successional dynamics of Gambel oak communities. This lack of information makes an assessment of HRV difficult and any such assessment can only be tentative and approximate. This lack of information also points to a need for research to be conducted in these communities. For the sake of the present assessment, we will draw on the few studies that have been conducted on Gambel oak disturbance regimes, even though they were located outside of GM.

Fire has long been considered the most important natural disturbance process in Gambel oak communities in western Colorado (Brown 1958). Gambel oak occurs in several types of communities including oak dominated shrubland and ponderosa pine-oak woodlands, only the former of which occurs in the GM. Each of these communities is likely to have had a unique pre-settlement natural fire regime. In Mesa Verde National Park (MVNP) the fire regime of oak-dominated shrubland has been described as one of moderate-size (*c.* 800 – 2000 ha) stand-replacing fires (Floyd et al. 2000). In contrast, the fire regime of pine-oak systems is likely to have been one of frequent low severity fires, as suggested by a study of such a system in Arizona (Fulé *et al.* 1997).

Conducting fire history studies in oak-dominated shrubland is difficult because none of the major species form fire scars. Therefore, studies must be based on the stand-origin method. Floyd *et al.* (2000) calibrated a method of reconstructing the fire regime of an oak-dominated community by comparing the interpretation of stand-origin dates with the occurrence of historically documented fires in MVNP. Since 1926, 90% of fires were caused by lightning and 90% of all fire events burned an area of < 0.1 ha. Only 2% of the fires burned an area > 4 ha, but these large fires accounted for > 95% of the total area burned in 20th century. All of these fires occurred in late June – early July, except for one which occurred in August, and all occurred under conditions of high wind and prolonged drought (Floyd et al. 2000). Based on an approximate

correspondence between the interpretation of stand-origin dates and the actual year of fire occurrence, Floyd et al. concluded that it was possible to reconstruct the fire regime of shrubland with decadal resolution to at least *c.* 1840, which predates Euro-American settlement of the area around MVNP. While the stand-origin method is likely to miss many small or low severity fires, it is adequate for reconstructing the occurrence of the severe fires that had a greater ecological impact and that largely shaped the present landscape.

The pre-settlement fire regime in MVNP was interpreted to be one of moderate-size (800 – 2000 ha) stand-replacing fires. A combination of fire suppression, grazing, and wet weather was probably responsible for a reduced occurrence of fire in the early 20th century. However, the fire regime in the latter half of the 20th century was again characterized by moderate-size stand-replacing fires. While several factors including fuels and topography affect fire regimes, the fire regime of MVNP was primarily controlled by weather. These fires occurred during extremely dry summers, when it was difficult to impossible to control the fires even with modern fire-fighting technologies. Given this type of fire regime in which severe and extensive fires occur infrequently in association with unusual fire-promoting weather, it is unlikely that the 20th century suppression of fires during years of less extreme fire weather has greatly altered the natural fire regime. As with pinyon-juniper woodlands, it is our interpretation that, while the behavior and effects of many or even most fires in Gambel oak shrublands in western Colorado may have been altered by fire suppression activities, the overall effect of fire suppression on ecosystem structure and dynamics has not been great. As further discussed in section 5.3, the modern fire regime of Gambel oak-dominated shrublands in GM probably does not differ greatly from the historic regime in terms of its overall ecological impact.

Aspen forests: At higher elevations in the GM, above the zone of Gambel oak, aspen becomes the dominant cover type (Figure 3.4c). Aspen can occupy a broad range of habitat types, varying from relatively xeric sites at lower elevation to more mesic ones at higher elevation. In other parts of the Colorado Rockies, such as the Front Range, lodgepole pine occupies habitats similar to those occupied by aspen and both species often play an important role in post-fire successional patterns to dominance by spruce and fir (Peet 2000). However, lodgepole pine is absent from GM. In the absence of this pine in GM aspen occupies a substantially larger portion of its potential range and is characterized by different successional patterns than elsewhere. Aspen is similar to lodgepole pine in its ability to dominate early post-fire stand development but it has substantially different life history traits. The underground portions of aspen clones are extremely long-lived and have been proposed to be among the oldest living organisms (Barnes 1966). Most clones of aspen in the Rocky Mountains are thought to have become established several hundreds or thousands of years ago, presumably during a time of more favorable climate, and to have persisted via asexual root sprouting (Mitton and Grant 1996). Following fire, aspen resprouts from its underground rhizomes and produces abundant, rapidly growing root suckers that favor the initial dominance of aspen (Peet 2000). Reproduction of aspen in western Colorado is overwhelmingly from suckering, but observations after the 1988 Yellowstone fires suggest that successful seed reproduction may depend on the coincidence of a wet spring, an absence of intense browsing, and good seed production during the year after a fire (Romme et al. 1997). Similarly, the establishment of aspen seedlings was observed in Routt and White River NFs following fires in 2002 (Kulakowski et al. *unpublished data*). Successful regeneration of aspen by resprouting also depends on conditions following the disturbance. For example, on the Uncompahgre Plateau aspen regeneration following clearcuts was inhibited on sites that had a high water table, heavy browsing, soils with a thin Mollic surface layer (as opposed to a thick Mollic surface layer), and compacted soils (Johnson 2000). Regeneration was especially inhibited at sites that had a combination of these characteristics.

Aspen stands typically have open canopies with relatively high light levels in the understory. Thus, spruce, subalpine fir, and sometimes even aspen are able to establish and grow up through the aspen canopy (Peet 1981). Aspen ramets (stems from root suckers) are often short-lived, rarely exceeding 200 years of age, so that the longer-lived conifers may eventually take over dominance of post-fire stands in certain areas. In such cases, spruce and fir often establish at a later stage of stand development, but sometimes may also initially co-dominate the site along with aspen. In some stands of western Colorado, including parts of

the GM, aspen establishment has been episodic and probably related to severe disturbances, such as fire (e.g. Figure 3.5; Kulakowski et al. 2004). Although fire is a likely explanation for this episodic establishment of aspen, pulses of establishment may have also resulted if mortality of canopy trees occurred coincidentally due to simultaneous mortality of aspen stems of an individual clone, resulting in stimulation of sucker re-growth. Such simultaneous mortality could be a consequence of cohort senescence for which there is substantial evidence from studies of aspen dynamics in Colorado and elsewhere (Kurzel, 2004). Drought

may be an important factor in contributing to widespread aspen mortality as appears to be occurring in the summer of 2006 in western Colorado. In the GM, the aggregated age structure of aspen stands from the FSVeg RMSTAND data base shows that most current stands established between 1880 and 1920 (Figure 3.6). This establishment is most likely to have been a response to the extensive fires of the late 19[th]century. In the same stands where aspen establishment is episodic, conifer establishment can be relatively continuous (McKenzie 2001, Kulakowski et al. 2004). This suggests that in such areas aspen
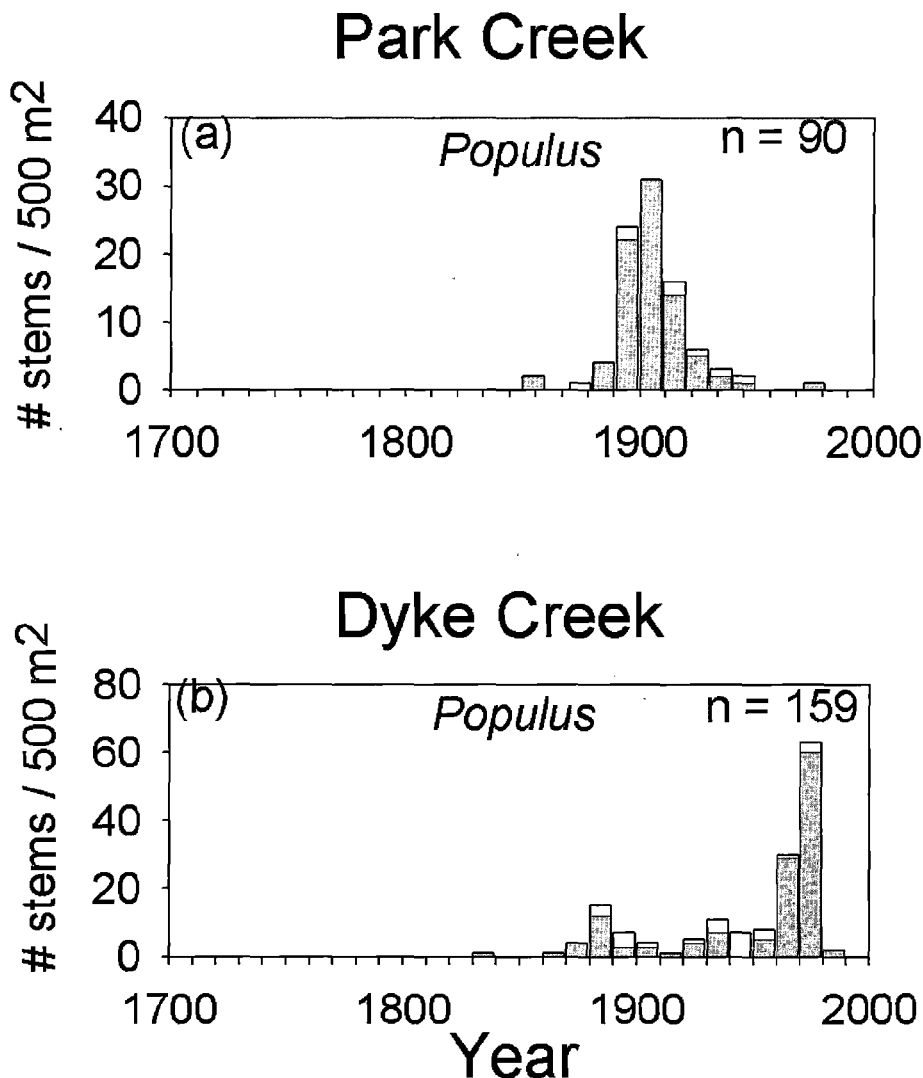
Figure 3.5—Dates of establishment of all successfully dated trees (> 3 cm dbh) in 10 year classes in two representative stands that were subjectively selected in the GM. Gray bars indicate date of pith or estimated date of pith (Duncan 1989), white bars indicate minimum date of establishment. Figure from Kulakowski et al.2004.
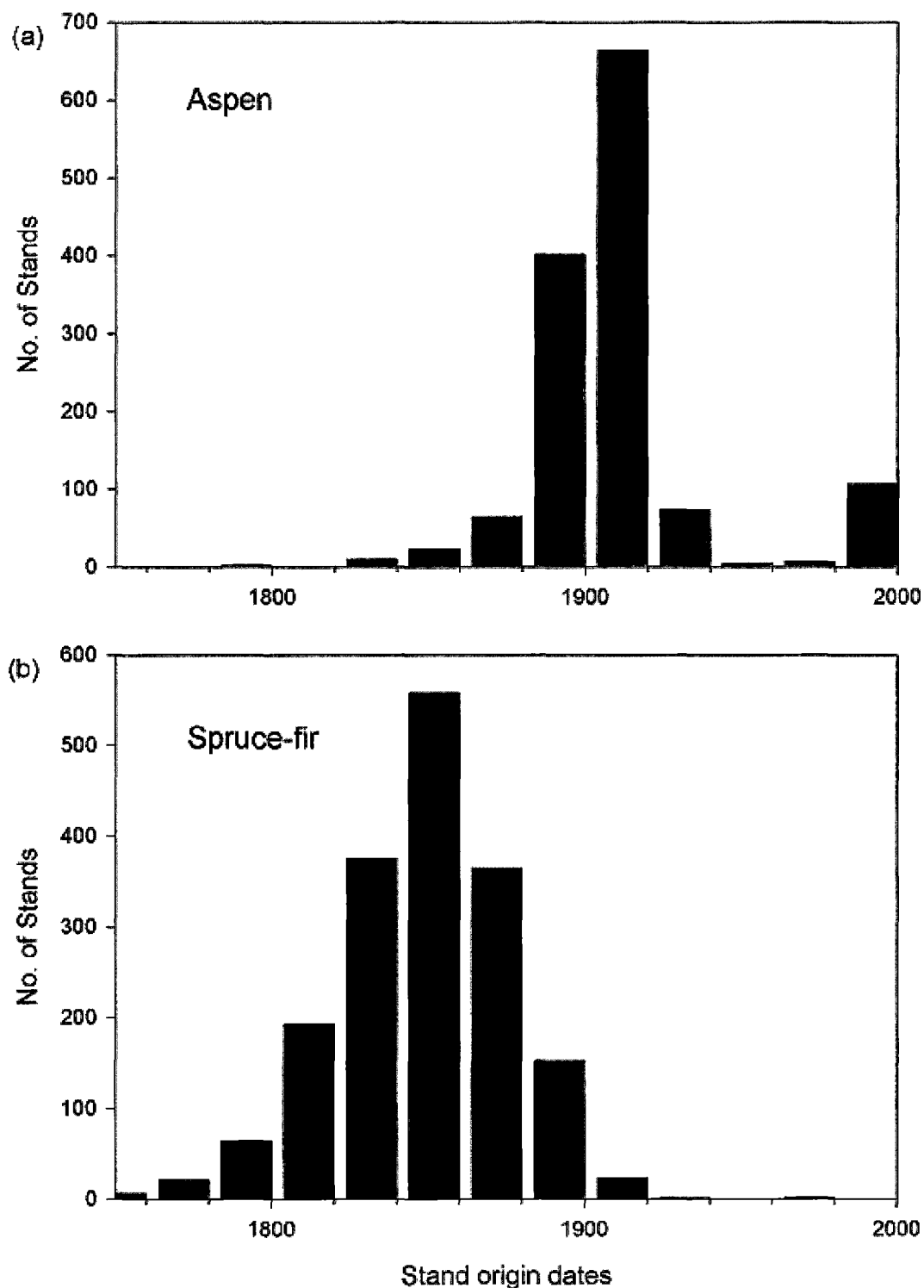
Figure 3.6—Stand origin dates in 20 year age classes as estimated from the FSVeg RMSTAND data base. Stand origin dates are based on the ages of the largest size class of trees that meet minimum requirements of dominance. Tree age is measured at breast height and a correction factor of 30 years is added for spruce, 15 years for fir, and 4 years for aspen.

require fire or other stand-replacing disturbance to persist.

Although seral to other forest types over much of its range, some aspen stands are self-maintaining (Langenheim 1962, Mueggler 1985, Peet 2000). For example, lower elevations that are transitional from steppe or shrubland to coniferous forest in the Southern Rockies are believed to support stable stands of aspen (Mueggler 1985, Peet 2000). At higher elevations in western Colorado, aspen have been suggested to be seral to the conifers, particularly at sites where pre-disturbance vegetation was dominated by spruce and fir (Langenheim 1962). This landscape pattern is also evident in the GM. Aspen appear to form self-replacing stands over the majority of its extent, particularly in the elevational range where site conditions may be more favorable for aspen than either Gambel oak or spruce and fir (Kulakowski et al.2004). Within its upper elevational range, aspen's seral vs. self-replacing status varies spatially, but aspen tends to maintain its relative successional status at a given site following disturbance (Kulakowski et al.2004). On some sites, aspen and the associated understory species may enhance the development of mollisols, which have a higher pH, more organic matter, and a greater cation exchange capacity, making a site less suitable for conifer growth and more favorable for aspen regeneration (Cryer and Murray 1992, Jones and DeByle 1985). Baker (1925) suggested that repeated fires could maintain stable aspen stands by eliminating seed sources for the conifers while promoting the suckering of aspen, however during our research in the GM area we have found no evidence of frequent fires near aspen stands in this area. Although edaphic factors are likely to play an important role in aspen's successional status at a given site, presently it is not possible to determine whether soil characteristics, initial floristic composition of a site, or other variables are the primary determinants of aspen's successional status at a particular site. However, research does show that aspen tends to maintain its relative successional status at a given site following disturbance (Kulakowski et al. 2004).

The relative abundance of aspen at a landscape scale is likely to be sensitive to any major changes in fire regimes. The occurrence of extensive and severe fires are likely to increase the extent of aspen, while a cessation of burning may allow the successional replacement of aspen by conifers at higher elevations. However, as discussed in section 5.3, the widespread belief that 20th century fire suppression has resulted in a decline in aspen in comparison to its late 19th century extent (e.g. Club 20 1998) is not supported for GM. We stress that current research does not suggest that present aspen extent is below historical levels in this part of western Colorado, or that associated dynamics are outside of their historical range of variability. These findings also do not support the notion that aspen is threatened because of fire exclusion or other management activities during the 20th century. Instead, in portions of the western Colorado landscape the extent of aspen may vary greatly at a centennial time scale in association with the infrequent natural disturbances that shape many of these forests. Where the successional replacement of aspen by conifers is occurring in such forests, such a trend may be within the range of historical variation.

It has been argued that browsing by native ungulate populations have resulted in a widespread decline in aspen extent during the 20th century (Kay 1997). The ability of native herbivores to seriously affect the regeneration of aspen is well documented at the scale of individual stands (DeByle 1985). At a regional scale it is uncertain, however, if elk populations significantly reduce the extent of aspen in the landscape. Studies in Rocky Mountain National Park have demonstrated that: 1) heavy use of some aspen stands by elk can seriously impede aspen regeneration (Baker et al. 1997); 2) that the degree of elk impact on aspen regeneration is highly variable over larger areas (Suzuki et al. 1999); and 3) over large areas the persistence of aspen is not threatened by current levels of elk populations (Shepperd et al. 2000, Kaye 2002)

Engelmann spruce-subalpine fir forests: Forests of spruce-fir dominate the highest elevations of GM (Figure 3.4d). These two species can co-exist in old stands not affected by coarse-scale disturbance for several hundred years (Veblen 1986b, Peet 2000). The major coarse-scale disturbances affecting spruce-fir forests in western Colorado are fire, spruce beetle outbreaks and blowdown. In addition to the patterns described above for post-fire development from dominance by shade-intolerant aspen toward dominance by Engelmann spruce and subalpine fir, the latter two species also regenerate following fires and can co-dominate the site from the time of stand initiation, especially at sites that lack seed sources of pines or root suckers of aspen (Rebertus et al. 1992, Peet 2000). Following fire, Engelmann spruce is likely

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to establish in greatest abundance, and there may be a lag in the establishment of fir. For example, age-structure studies have shown that fir establishment typically lags that of spruce by several decades (Whipple and Dix 1979, Veblen 1986a, Rebertus et al. 1991); however, where seeds are available, both species may regenerate immediately following fire (Doyle et al. 1998). The reestablishment of spruce and fir following fire is significantly affected by the size of the burn, by the availability of seed, and by climate (Peet 1981, Alexander 1984, Tomback *et al.* 1990, Rebertus *et al.* 1992).

At some sites, bimodal age distributions of spruce indicate both an immediate post-fire cohort and a second cohort that establishes c. 150 to 250 years after the initial cohort begins to thin (Aplet et al. 1988). At other, apparently less favorable, sites where canopy closure is less complete, bimodal age distributions of Engelmann spruce are not found (Whipple and Dix 1979, Veblen 1986a, Veblen et al. 1991a, Rebertus et al. 1992). Such variation in age structures with site conditions is an illustration of the importance of site variation to stand development patterns in the subalpine zone (Peet 1981, Veblen 1986a).

Given the long period of post-fire stand development before the initial post-fire colonists die (> 500 years), there is a high probability that the stand will be affected by a major canopy disturbance in the form of a windstorm or spruce beetle outbreak (Veblen et al. 1989, 1991a, 1991c, Eisenhart and Veblen 2000, Kulakowski and Veblen 2003). These large-scale canopy disturbances typically reduce the relative dominance of the main canopy by spruce, and result in vigorous releases of subcanopy individuals of both species (Schmid and Hinds 1974, Veblen et al. 1989, 1991c, Kulakowski and Veblen 2003). In old stands, in gaps created by the death of a single tree or small groups of canopy trees, both species regenerate (Veblen 1986b, Veblen et al. 1991a).

3.4. Human Settlement and Land Use

3.4.1. The Native American Period

Native Americans have used the area of the GM for thousands of years. There has been much philosophical debate regarding the definition of "natural" and the severity of Native American impact that would result in a significantly modified or "unnatural" landscape (Vale 1998, Denevan 1992).

Rather than resolving this debate or being concerned with implied "naturalness", our aim is to understand the relative roles of non-anthropogenic disturbances, disturbances caused by Native Americans, and disturbances caused by modern humans in shaping the landscape. In other words, for particular habitats in GM did Native American land-use practices and potential impacts on natural disturbance patterns have an ecologically significant impact on vegetation structure and/or composition? Western Colorado's first human residents arrived approximately 13,000 years ago and subsisted primarily by hunting megafauna and, in certain environments, by gathering plant products (Reed and Metcalf 1999). The Clovis Complex is the oldest of the Paleo-Indian divisions in North America and affected large areas of North America, including parts of western Colorado (Cassells 1997, Reed and Metcalf 1999). In comparison with plains peoples, subsistence of foothill and mountain peoples relied more on medium-sized game that were killed individually, rather than en masse, and on gathering plants. Artifacts from this Paleo-Indian era have been found in the area of present-day GM (Cassells 1997, Reed and Metcalf 1999). Evidence of early settlement and hunting in the GM area suggests that this region was less intensively utilized than in later periods. In fact, in most of western Colorado, early Paleo-Indian populations were low and use of this area was sporadic. Furthermore, many high elevation sites were probably uninhabitable due to glaciation and/or severe weather, especially during winter.

Climate warmed during the mid-Holocene and native populations became less nomadic and a greater number of established settlements appeared in western Colorado (Reed and Metcalf 1999). This Archaic period, which lasted approximately between 6400 B.C. and 1 A.D., was associated with more systematic use of local resources and increased seasonal use of higher elevations. The Fremont culture probably settled portions of western Colorado around 1 A.D. and relics of this culture has been found in the GM area. The Fremont people were concentrated in western Rio Blanco and Moffat Counties, the Glade Park area southwest of Grand Junction, and western Montrose County. This culture relied more heavily on horticulture, especially corn, than its predecessors, but hunting and gathering were still important. There was also an increase of permanent structures and small villages during the Fremont period.

By the 13th century, peoples of the Fremont culture as well as the Anasazi Culture of western Colorado exited Colorado, contracted geographically, or altered their lifestyles to return to older forms of hunting and gathering without agriculture (Wyckoff 1999). The ensuing Protohistoric era was thus characterized primarily by hunting and gathering (Reed and Metcalf 1999). Seasonal patterns of movement were similar to those during the Archaic period and pinyon and juniper woodlands may have been especially well suited to winter habitation. During spring, summer, and fall, groups utilized the high country more, hunting deer, elk, and pronghorn. Utes occupied western Colorado since at least the early 1600s (Casells 1997). Ute contact with Spaniards began in the early 1600s and resulted in an equestrian lifestyle for the Utes (Reed and Metcalf 1999). By the late 18th century, the Ute were primary occupants of western Colorado, although the Shoshone may have occupied the northwestern part of the state (Reed and Metcalf 1999). The Utes were generally nomadic, and their travels were shaped by the annual cycle of seasons and game movement. Hunting and gathering activities provided the majority of the Ute's diet, and they only occasionally cropped maize or beans (Hughes 1987, Crum 1996, Reed and Metcalf 1999). In the areas of the present-day GM, Ute modification of the landscape through agricultural practices was probably very slight and spatially limited to riparian habitats at low elevation. They also collected plants such as yucca, yampa roots, grass seeds, pinyon nuts, and berries.

Native American activities such as hunting, plant collecting, agriculture, and burning undoubtedly had local impacts on the landscape of the Rocky Mountains, but the extent of these impacts is largely unknown. Fire was potentially the most effective tool for landscape alteration by Native Americans, but the extent to which intentionally set fires increased the area that otherwise would have been burned by lightning-ignited fires is impossible to determine. Even the size of the Native American population in the Rocky Mountain region at the time of the earliest contacts with Spaniards in the 16th century is unknown (Thornton 1987). Estimates of the size of the Native American population north of Mexico are highly speculative and cover a wide range (Thornton 1987, Denevan 1992). Simmons (2000) gives a range of 5,000 to 10,000 for the maximum Ute population over their entire range in present day Utah, western Colorado, and northern New Mexico at the time of first

white contact. Even allowing for substantial population decline during the 16th to 19th centuries due to the ravages of introduced European diseases, the Ute population in the GM still was small. Given the low population densities of the Native American inhabitants of the Rocky Mountain region (Thornton 1987, Simmons 2000), it is primarily through the use of fire that they could have had a significant impact on the landscape. Hunting was a motive for Native Americans to burn the vegetation of some habitats in the Colorado Rockies. Fire was widely employed by Native Americans in North America as a tool to drive game and to attract game through improved forage (Stewart 1956, Pyne 1982). Fires also may have been set in the Rocky Mountain region for warfare, communication, clearing of travel routes, clearing of tall vegetation that could conceal enemies around campsites, and to improve grazing for horses (Barrett 1980a, 1980b, Gruell 1985, Burns 2004, Raish et al. 2005). These uses of fire are derived largely from early historical accounts by Euro-Americans and Native Americans, but how frequently or extensively human-set fires affected particular sites is unknown.

In the 1930s, Omar Stewart (1942) systematically interviewed elders from 14 bands of Utes from Utah, New Mexico, Arizona and Colorado and gathered important information on the use of fire by the Utes. Eight of the bands formerly hunted in Colorado. These Ute elders reported that during the 19th century their hunting practices included the use of fire to drive antelope, rabbits, deer, and elk. The most commonly reported use of fire (i.e. by 7 of the 8 bands represented by the informants) was for driving rabbits out of thick brush and encircling them (Stewart 1942). Given the distribution of rabbits from the Plains to alpine treeline, such use of fire potentially was widespread in Colorado.

The effects of fires ignited by the Utes are likely to have varied according to habitat. For example, in grassland and shrubland habitats fuels dry out more easily than in dense forests, and such habitats were favorable to certain game animals (e.g. rabbits, antelope). It is likely that at lower elevations, human-set fires could have easily spread during most years because of consistently dry summers. Human-set fires may have been more frequent near the ecotone between forests and low elevation grassland both for reasons of fuel desiccation and game habitat. In contrast, in the subalpine zone, the fire regime is characterized by infrequent-stand replacing fires, and

many decades pass before fuel and or weather conditions permit fire recurrence to the same stand of 100s or 1000s of hectares (Veblen 2000, Sibold 2001, Kulakowski and Veblen 2002, Veblen 2003, Sibold et al.2006, Buechling and Baker 2004). Thus, even if Native Americans ignited fires frequently in the subalpine zone, the tree-ring evidence of fire indicates that fires only spread infrequently. Given that occurrence of widespread fire in the subalpine zone is not limited by ignition (See Section 5.3), but rather is dependent on relatively rare drought conditions, it is unlikely that Native Americans had a major impact on the fire regime of these high elevation forests.

Intentional burning by Native Americans as a means of harassing white intruders during the 19th century was reported in many areas of Colorado. Early settlers in the Front Range and in northwestern Colorado attributed many fires to purposeful or accidental acts of Native Americans, and many of these attributions later were reported as hearsay in government reports on early forest conditions (Jack 1900, Sudworth 1900). Extensive burning in the late 1840s in the Pikes Peak region was attributed by settlers in the 1890s to Native Americans who were attempting to drive the game out prior to the expulsion of the Native Americans from the area (Jack 1900). Both Sudworth (1900) and Jack (1900) noted, however, that early settlers sometimes attributed fires to Native Americans when they themselves had set the fires, and recently Baker (2002) also has questioned the reliability of early settlers' reports of fires set by Native Americans. However, we do not have any objective basis for assuming that all early reports of fires set by Native Americans were false.

An increase in fires set by Native Americans may have resulted from the intensifying struggle between white settlers and Utes for control of the land in northern Colorado in the 1870s (Simmons 2000). The Yampa Utes lived primarily between the Yampa and White Rivers but hunted widely in Flat Tops, Middle Park, and North Park. In the 1860s, the U.S. government created an Indian Agency near Meeker on the White River to which the Yampa Utes were assigned. Not all the Yampa Utes had agreed to removal to this reservation, and those who had grew increasingly angry over the Indian Agency's failure to comply with the terms of treaties. The Utes never conceded that they had given up hunting rights on lands outside the reservation, and in the mid-1870s were increasingly angry over competition from livestock on their hunting grounds in Middle Park. They were further outraged by the prohibition against their use of their traditional hot springs at Hot Sulphur Springs (Wier 1987, Simmons 2000). During the late 1870s settlers argued that the Utes were burning forests in the Middle Park, North Park and Yampa Valley areas as an angry gesture against the whites.

Yet, the Utes complained to the Governor of Colorado that the charges against them were false, and at least one white observer noted that some of the fires had been started by whites (Wier 1987). George Grinnell, a traveler, passed through North Park and into Middle Park in late summer of 1879. He claimed that some of the fires in North Park were set by the ranchers to clear land of the sagebrush (Wier 1987).

The Ute chiefs also denied that their people were responsible for intentional burning (US House of Representatives 1880). In 1880, the chief of the Utes, Ouray, testified to the US Congress in response to an inquiry into Ute burning of forests and houses:

> I have heard nothing of it except what I have seen in the papers.... Last year was a very dry year. Where there had been a camp over night and a little fire was left, in the morning a wind might have come up and scatter it and set fire to the place, and the whole country would burn up in that way. I do not think anyone was to blame - miners, campers, or anyone else. It was so dry that there was no grass. Everything was dry and dead, so that it was easy for everything to catch fire.

In the area of present day GM, unlike other parts of western Colorado, the extensive burning during the late 19th century was not attributed to the Utes (Sudworth 1900). Rather, the fires were reportedly started primarily by neglected camp fires, coal mines, and cowboys who tried to improve some of the land for cattle grazing. Thus, there are claims that both Utes and white settlers set fires in the late 19th century, and, of course, some of the fires may have been ignited by lightning.

Although there is uncertainty about the sources of ignition, there is no doubt that a vast area of forest was burned in western Colorado in the late 19th century. In addition to the documentary historical sources, tree-ring dating of fires in the Park Range confirms that 1879 was a year of extensive burning (Bartlett-Howe 2001, Kulakowski and Veblen 2002). Numerous

widespread fires occurred in northern and central Colorado in the late 19th century (Jack 1900, Sudworth 1900, Kipfmueller and Baker 2000, Veblen et al. 2000, Sibold 2001, Kulakowski and Veblen 2002, Kulakowski et al. 2003, Sibold et al.2006). As discussed below, according to tree-ring records of climatic variation this period included several exceptionally dry years (Cook et al. 1998). Even if some of the widespread fires of the late 19th century were ignited by humans (and there is no certainty on this issue), it is likely that fires ignited by lightning would have had approximately the same impact on the landscape during these exceptional droughts. The legacy of post-fire stands of aspen and spruce-fir from the fires of 1879 and the 1880s is still a dominant feature of the landscape of northern Colorado. Stands originating from these burns show a significantly lower susceptibility than older stands to subsequent disturbance by windstorms (Kulakowski and Veblen 2002) and spruce beetle outbreaks (Kulakowski et al. 2003, Bebi et al. 2003). Large disturbances occur infrequently in Colorado subalpine forests, but they exert an important influence on the patterns and processes of these ecosystems. As an example, the extensive burning in the subalpine zone of western Colorado in 1879 and the 1880s caused long-lasting changes in the stand structure and composition of much of this landscape. In turn, these changes in structure and composition affect stand susceptibility to subsequent disturbances.

Overall, for GM there is no compelling evidence that Native Americans had a major influence on the vegetation of the subalpine zone through intentional burning or other activities. Even at lower elevations, where human use of the landscape was probably more frequent it is likely that Native American impacts through burning and/or plant collecting activities were relatively minor and probably limited to particular habitats.

3.4.2. The Euro-American Period

Permanent Euro-American settlement in the Colorado Rockies is usually dated from the mineral discoveries of the late 1850s. Earlier Euro-American travelers also passed through the area that was to become the GM. In 1776, the Domínguez-Escalante expedition traveled through the area but their journal does not report any major nearby white settlement nor impacts on the landscape (Chavez and Warner 1976).

Although their journal's description of the forested landscape is not detailed, it does describe a landscape that is compositionally similar to that of the present day, including oak and mixtures of conifers and aspen. French and American fur traders were active in Colorado in the eighteenth century, but their activity in the GM area is not known.

The earliest detailed published description of GM was by George B. Sudworth (1900), a surveyor of the U. S. G. S. who traveled through and mapped the Battlement Mesa Forest Reserve in 1898. Sudworth reported that very little land had been settled within the Reserve and that he encountered only a few tracts of abandoned agricultural and ranch land. Mining and related activities had a more substantial impact on the landscape around the GM during the second half of the 19th century. Mining was often damaging to riparian habitats in Colorado by both alterations of the water courses and by discharge of pollutants and sediments (Wyckoff 1999). Sudworth reported the presence of extensive coal mining around the Reserve with heavily logged forests, but only on a local scale. Mining operations were also partly blamed for the widespread fires of the late 1800s.

Large herds of cattle ranged unchecked and uncounted across the Colorado high country throughout the latter half of the 19th century. Over grazing was widely perceived to be degrading the high country of Colorado, and was one of the motivations for creating federal timber reserves in the 1890s to 1905 (Wyckoff 1999). Although exact numbers are not available, the density of cattle was reportedly higher in GM than in other areas of western Colorado, such as the area of present-day White River National Forest (Sudworth 1900). The probable impacts of grazing are discussed in section 6.2.3. The late 19th century was also a time of widespread construction of reservoirs and ditches on GM to supply water to the surrounding agricultural areas. The potential ecological impacts of these earthworks are discussed in section 6.2.2.

While widespread timber harvesting began in much of Colorado in the 19th century, the GM area was generally less affected, except for areas immediately surrounding coal mines near the Reserve (Sudworth 1900). By early accounts, the timber in the GM was "exceedingly poor -- second- or third-rate, from which almost no clear lumber could be expected" (Sudworth 1900). Very little of any tree species was cut by early settlers and, of what timber was used, the dead timber

was preferred because of its lighter weight. There is no account of the amount of timber exploitation during the first half of the 20th century in the GM.

Recognizing the need to preserve timbered lands in the West, a reserve system was established by Congress in 1891 (Wyckoff 1999). Grand Mesa National Forest originally was part of the Battlement Mesa Forest Reserve established in 1892. In 1907 this area became the Battlement Mesa National Forest and in 1916 it was divided into the Grand Mesa National Forest, while the eastern part of this area became part of the Gunnison National Forest. The chief goals of the reserve and national forest system were to protect natural resources from over-exploitation and to protect forests from wildfire (Ingwall 1923). Active fire suppression began with the creation of the reserves, but it was not until the early 1910s to 1920s that sufficient personnel and equipment were available to monitor and extinguish fires.

4. Climatic Variation

Climatic variation can be a critically important influence on tree recruitment, growth, and mortality patterns (Brubaker 1986, Auclair 1993, Woodward et al. 1995, Savage et al. 1996, Villalba and Veblen 1998, Pedersen 1998, Breshears et al. 2005) as well as on disturbances such as fire and insect outbreaks (Swetnam and Lynch 1989, 1993, Swetnam and Betancourt 1998). Consequently, we briefly review climatic variation, especially over the past c. 500 years, as a background to discussing the possible effects of climatic variation on disturbances in Chapter 5 and vegetation changes in Chapter 6. We first discuss year-to-year (high-frequency) climatic variation and then long-term trends in climate (low-frequency variation).

Patterns of inter-annual climatic variability that may affect ecological processes in the GM are often related to variations in broad-scale ocean-atmosphere interactions such as the North American monsoon and teleconnections of weather patterns associated with variations in sea surface temperatures in the tropical Pacific, the North Pacific and the North Atlantic (Adams and Comrie 1997, Higgins et al. 1998, Carelton et al. 1990, Gray et al. 2003). From southern to northern Colorado there is a declining influence of the North American monsoon, as reflected in the south-to-north gradient of decreasing precipitation peaks in July and August associated with the summer monsoon

(Donnegan 2000). The periodic occurrence of anomalous oceanic and atmospheric conditions in the tropical Pacific known as El Niño Southern Oscillation (ENSO) influences temperature and precipitation conditions in Colorado. Although variable over multi-decadal time scales, the short-term periodicity of ENSO events is approximately 2-6 years (Michaelsen and Thompson 1992, Swetnam and Betancourt 1998). ENSO events exhibit a phase-locking, where positive (cold, Niña) and negative (warm, Niño) events tend to follow one another, with quick shifts in phases occurring typically within 3-4 months (Diaz and Kiladis 1992). During warm El Niño events in the U.S., the mid-latitude winter storm track is typically located further south than during non-El Niño years (Swetnam and Betancourt 1998), steering greater amounts of precipitation, and often warmer temperatures, toward the southern U.S. Broad-scale analyses of the Rocky Mountain region from Idaho to southern Colorado show weak but significant correlations of snowpack to ENSO activity (Changnon et al. 1990). In Colorado, spring precipitation tends to be above average in association with El Niño events and below average in association with La Niña events (Diaz and Kiladis 1992, Woodhouse 1993, Veblen et al. 2000). Variations in sea surface temperatures in the North Pacific occurring at a mode of 20-30 years (i.e., the Pacific Decadal Oscillation; PDO) also influence the climate of the southern Rockies. Greater precipitation in all seasons in the central and southern Rockies is associated with the warm phase of the PDO (Gray et al. 2003). Severe drought conditions (e.g. 1950s) in the southern Rockies may ensue from coupling of the cold phase of PDO (e.g. 1947-1976) with unusually warm sea surface temperatures in the North Atlantic (e.g. 1930-1960, Gray et al. 2003). The importance of 60 to 80 year periods of warmer sea surface temperatures in the North Atlantic is now known to be associated with multi-decadal droughts in the western U.S., including Colorado (McCabe et al. 2004). Drought associated with warmer than average North Atlantic sea surface temperatures is strongly reflected by increased fire occurrence in the subalpine zone of the Colorado Front Range (Sibold and Veblen 2006), and 60-80 year periods of increased or decreased fire occurrence in western Colorado forests also appears to be associated with North Atlantic sea surface temperatures (Gray et al. 2003, Schoennagel, Veblen, Kulakowski, and Holz, unpublished data).

Climatic variations at time scales of 60 to 80 years

and longer are likely to have affected not only fire regimes, but also demographic processes of the plants that dominate different vegetation types probably resulting in shifts in ecotones. For example, at longer time scales, the Little Ice Age (c. 1550 to 1850 A.D.) was a period of climatic variation that potentially affected the locations of ecotones between many vegetation types in the GM region. This period corresponded to a period of generally cooler and drier conditions terminating in the mid-19th century that affected large parts of North America and other continents. However, because this time period of anomalous climate is not synchronous globally and is characterized by a wide range of climatic anomalies, it is better to refer to it as the 1600 to 1850 period of anomalous climate rather than as a Little Ice Age (Hughes and Funkhouser 1998). In the southern Rocky Mountains, this anomaly is reflected in lower annual precipitation as reconstructed from tree-rings in the Southwest (Grissino-Mayer 1995), the advance of glaciers in the Colorado Front Range (Benedict 1973, Grove 1988), and in fossil pollen assemblages from southwestern Colorado (Fall 1988, Petersen 1994). This time period includes the late 16th century period of megadrought which is strongly reflected in tree-ring records from the central and southern Rockies (Gray et al. 2003). However, the timing and nature of these climatic variations and how they may have differentially affected high versus low elevation habitats in Colorado are not well resolved. Nevertheless, changes in climatic conditions in the early to mid-19th century may have contributed to ecological changes at this time in Colorado.

Documentary evidence shows that during the reference period for this assessment of historic range of variability, western Colorado was affected by severe drought in the mid-19th century (US House of Representatives 1880, Sudworth 1900). Several recent tree-ring based reconstructions of droughts and stream flows further document extensive and often severe droughts in the central Rockies east of the GM (Woodhouse 2001, Woodhouse and Brown 2001). Severe droughts also appear to have occurred once or twice a century over the past 400 years in the central U.S. (Woodhouse and Overpeck 1998). Such lines of evidence suggest that severe droughts have likely been influential in forest development during the reference period.

Recent (i.e. 20th century) climatic variation in Colorado needs to be considered in relation to global warming. At a global scale there is abundant evidence of warming during the 20th century (Intergovernmental Panel on Climate Change 2000; Crowley 2000). Although global climate models generally agree in predicting further global warming, there are great uncertainties about how climates of particular regions will vary in future decades (Kerr 2000, Couzin 1999). At the broad-scale of the Colorado Rockies (essentially all of western Colorado), analysis of the regional 100-year instrumental climate records indicates no trend in mean annual temperature or precipitation (Kittel et al. 1997). This pattern for the Colorado Rockies contrasts sharply with the patterns for the Rockies of Montana and Wyoming where there are statistically significant trends of warming. In different regions of the Colorado Rockies, temperature and precipitation trends are complex due to variables which include mountain topography and possible influences of land use/land cover changes from the Plains through upslope advective transport of water vapor to higher elevations (Chase et al. 1999, Williams et al. 1996). Consequently, potential differences in trends must be considered for different elevational zones and locations relative to the zone of dramatic land use change that has occurred over the past 100 years.

To characterize climatic trends in the GM, spring and summer temperatures and precipitation were used instead of annual averages because of the greater influence of growing season climate on ecological processes such as tree growth and fire (Figure 4.1). Departures (standard deviations) from mean spring (March-May) and summer (June-July) temperature and precipitation were computed for five areas near GM over most of the 20th century: a) Bonham Reservoir, high elevation (precipitation only), b) Cedaredge, c) Collbran, d) Grand Junction, and e) Palisade (Figure 4.1). These precipitation and temperature records were selected as representative of trends in different geographic and elevational zones after extensive screening of all weather records from the GM region available from the Colorado Climate Center of Colorado State University. Many records were rejected due to high percentages of missing monthly data. The climate record was constructed by normalizing each station's record as departures from the mean, and then averaging the departures from the mean for each month across all stations through time. A 15-year centered-average smoothing curve shows long-term trends as compared to the mean (horizontal line) for the entire time period (Figure 4.1).
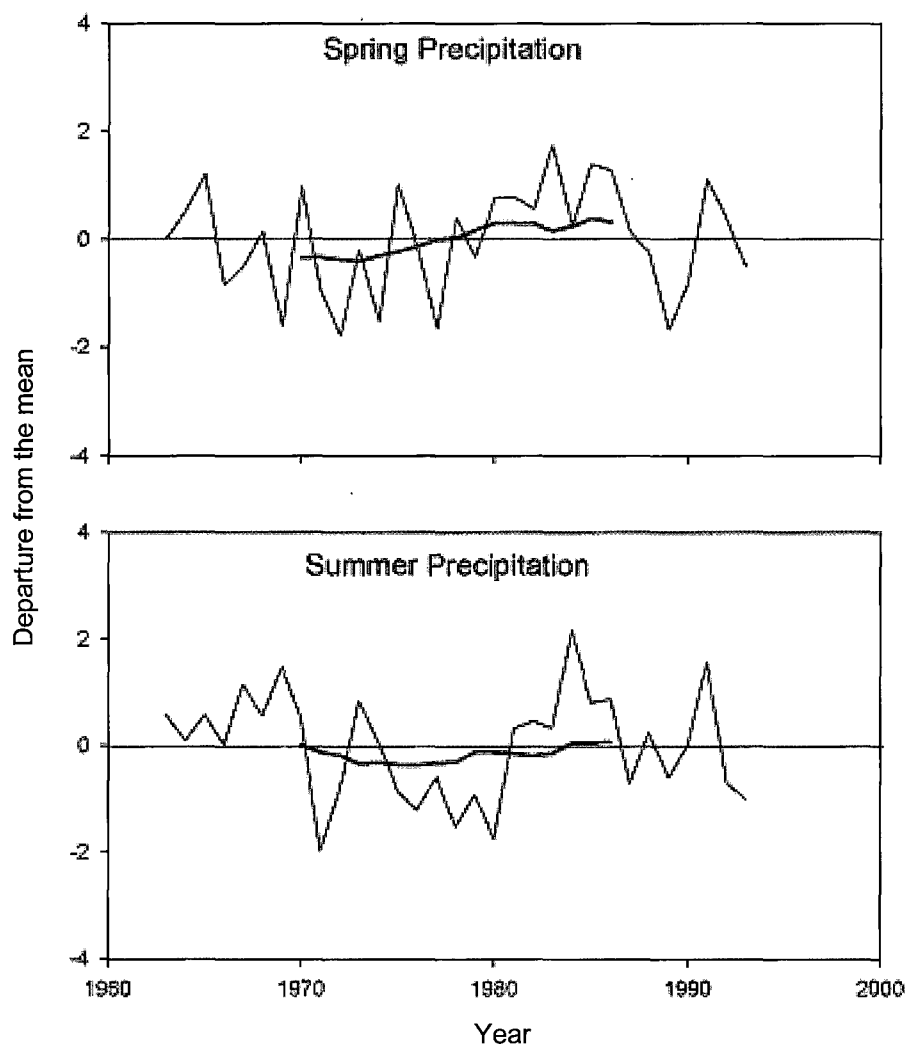
Figure 4.1—Trends of departures (standard deviations) from mean spring (March-May) and summer (June-July) temperature and precipitation for: (a) Bonham Reservoir; (b) Cedaredge; (c) Collbran; (d) Grand Junction WS; and (e) Palisade. See Table 3.1 for climate stations used in each regional record. A 15-year centered-average smoothing curve shows the long-term trend against the mean (horizontal line) for the entire time period.

Fig. 4.1a – Bonham Reservoir

Several consistent patterns are evident among the climate stations. Cedaredge, Collbran, and Grand Junction all show a cooler period in the 1910s and slightly elevated temperatures during the late 1930s to early 1940s. In Colorado, this shift from wetter, cooler conditions in the early 1900s to a more arid climate after 1932 has been previously noted and is evident in tree-growth patterns in the Front Range (Graybill 1989) and in streamflows in the upper basin of the Colorado River (Stockton 1976). These stations also show a trend towards increased spring precipitation since c. 1976 following generally below average precipitation from c. 1950 to the mid-1970s. This trend is consistent with the persistence of a cool (negative) PDO phase from 1947 to 1975 and a warm

(positive) PDO phase from 1977-1998 (Gray et al. 2003). Potentially, the ending of the warm PDO phase in 1998 has contributed to the drought conditions at the beginning of the 21st century. From the perspective of understanding how climatic variation may affect ecological processes such as fire and insect outbreaks, these seasonal patterns of temperature and precipitation indicate that: 1) linear or step-wise changes in multi-decadal scale seasonal climate are not evident during the 20th century; 2) there has been a great deal of variability in seasonal climate which would be expected to influence ecological processes at the scale of individual years to decades; and 3) for different areas in the GM region at different elevations, at the scale of individual years to a decade, seasonal trends
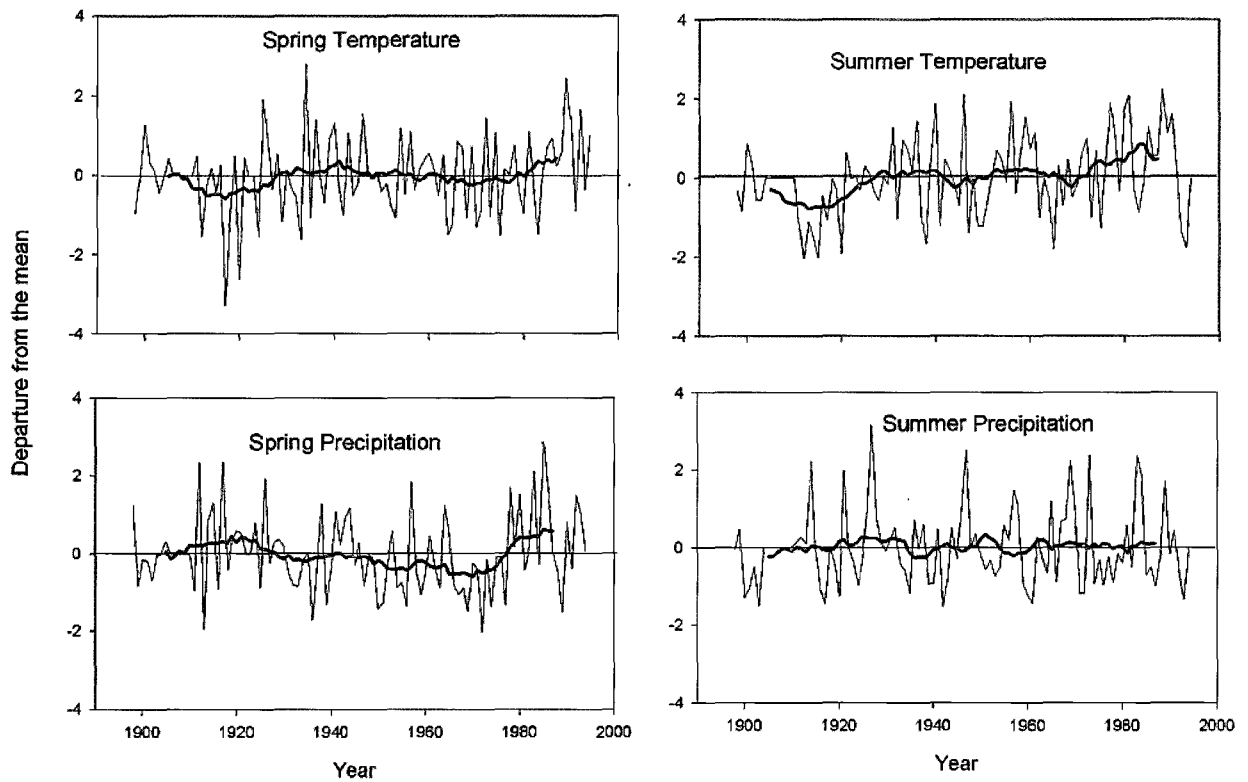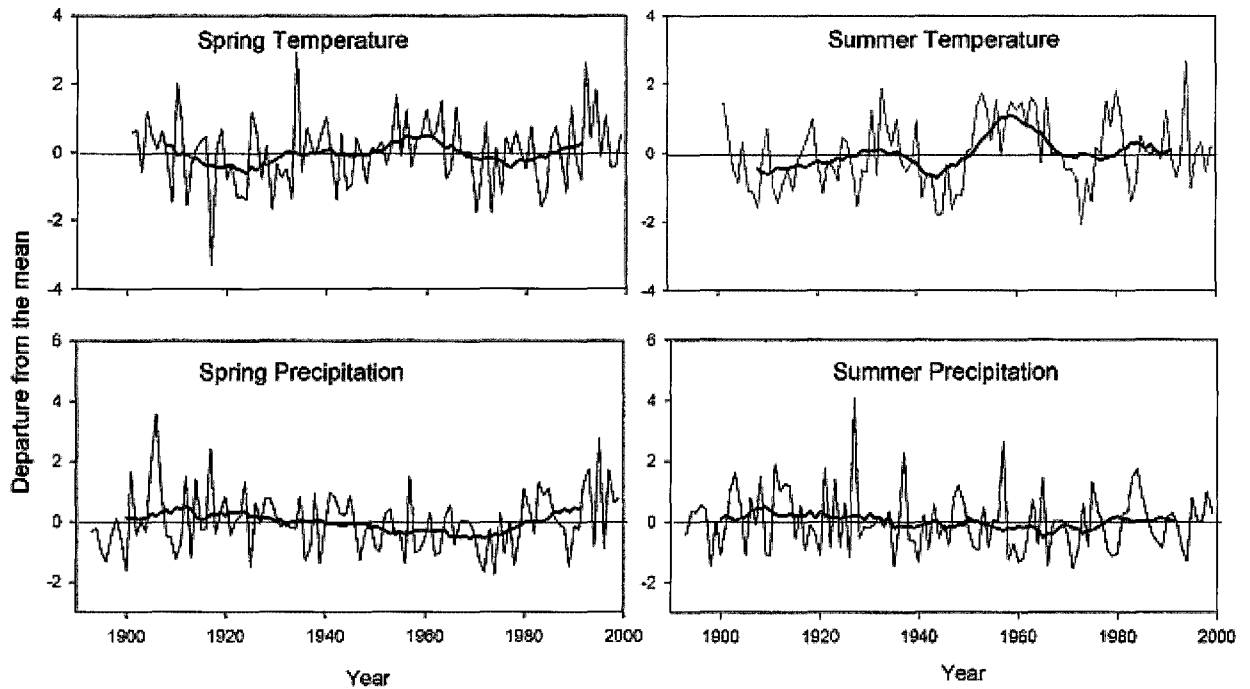
Fig. 4.1b – Cedaredge
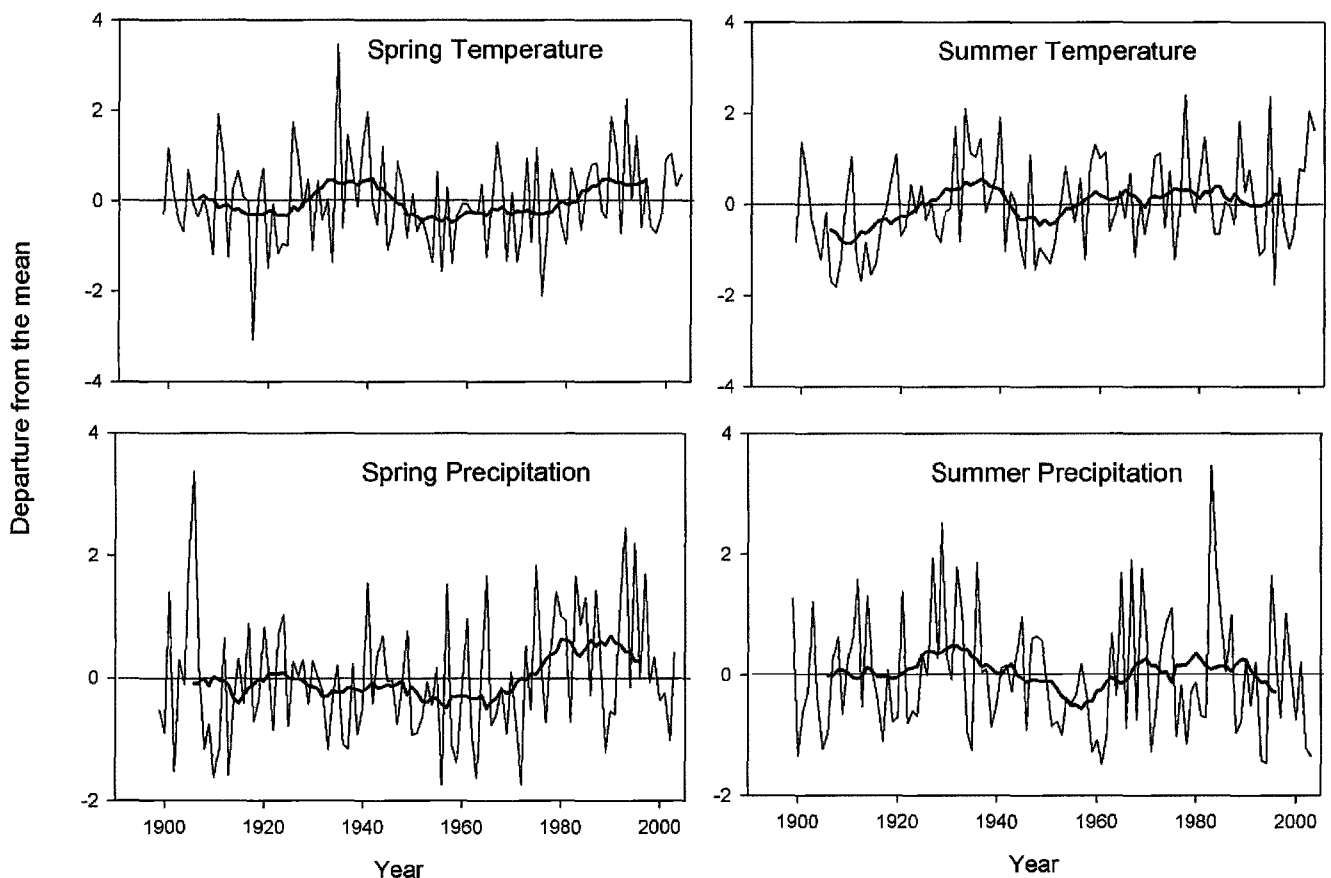

Fig. 4.1c – Collbran

Fig. 4.1d – Grand Junction WS

are not always in phase and sometimes show opposite extremes. Thus, in interpreting trends in ecological processes such as tree growth, fire, and insect outbreaks which are sensitive to climatic variation, different temporal patterns are likely for high- versus low-elevation habitats.

5. Disturbance Processes and Patterns

The following review of disturbance patterns and their ecological consequences is organized by type of disturbances (e.g., wind, fire, insects) and particular forest type. Wherever possible, the discussion will focus on studies and data from the GM. When no data or studies are available from the GM, the discussion will necessarily have to be based on information from nearby areas in the Rocky Mountain region.

5.1. Wind

In the lower elevational zones, pinyon pine and Gambel oak are rarely affected by severe blowdown. Pinyon pine's extensive root system and short stature may contribute to increases resistance to wind damage. The connection of roots underground may have a similar effect for oak. Furthermore, while strong winds occasionally do occur in this zone, these areas experience overall weaker winds and fewer extremely intense windstorms than forests at higher elevation. Blowdowns are rare in this elevational zone and tend to be small in area. For example, no evidence of widespread wind damage is evident in Pinyon pine woodlands in the nearby Gunnison Gorge area (Eisenhart 2004). Therefore wind storms apparently have only minor influences on stand structures in these vegetation types.

Exceptionally strong wind storms occasionally

cause extensive blowdowns in subalpine forests of the southern Rocky Mountains and are important determinants of stand development patterns (Alexander 1987, Veblen et al. 1989). For example, in 1987, a tornado blew down 6000 ha (14,820 ac) of forest in the Teton Wilderness (Fujita 1989, Knight 1994). In 1997, easterly winds of 200-250 km/hr blew down over 10,000 ha (24,700 ac) of forest on the western slope of the Park Range in Routt NF (Flaherty 2000). Blowdowns of moderate size (*e.g.* < 100 ha; 250 ac) have also been documented in western Colorado (Kulakowski and Veblen 2003). In addition to these rare but spectacularly large blowdowns, small blowdowns of approximately a fifth to several hectares are common in the subalpine forests of Colorado (Alexander 1964, Veblen et al. 1991a).

Two of the most common subalpine tree species (Engelmann spruce and subalpine fir) are shallow rooted and not windfirm, whereas aspen tends to be more resistant to wind damage (Veblen *et al.* 2001). Susceptibility to damage from severe wind storms varies with forest cover type and stage of post-fire stand development. For example, extensive analyses of damage patterns associated with the severe 1997 Routt blowdown (Baker et al. 2002, Kulakowski and Veblen 2002) show that: 1) younger stands (i.e. > *c.* 120 yrs old), which were more recently affected by stand-initiating fires, were less affected than older stands; and 2) aspen-dominated stands are less affected than spruce-fir stands. Climate conditions in the subalpine zone also contribute to windthrow. High wind events may be more common, rugged terrain may create greater turbulence, and there is higher probability of heavy snow loads (Alexander 1987). In this zone, windthrow is greater where topographic or logging patterns constrict and therefore accelerate winds (Alexander 1964). Other features which increase the hazard of windthrow in relation to cutting operations include shallow or poorly drained soils, location on leeward cutting boundaries, dense stands, infestation by root and butt rots, and steeper slopes (Alexander 1964).

Time since last fire, and therefore stage of seral development, is an important determinant of the successional consequences of disturbance by blowdown. For example, in 1973 a blowdown of a 350-year old post-fire forest in Rocky Mountain National Park accelerated succession from lodgepole pine towards subalpine fir and Engelmann spruce (Veblen et al. 1989). In contrast, in 1934 a blowdown

of an old-growth forest in the Mount Zirkel Wilderness changed forest dominance from Engelmann spruce to subalpine fir (Kulakowski and Veblen 2003). Comparison of wind disturbances in spruce-fir stands > 400 years old and an adjacent c. 250-year old post-fire stand revealed the latter to be less susceptible to small blowdowns (< 0.3 ha) (Veblen et al. 1991a). When blowdowns shift species dominance from spruce to fir (*e.g.* Kulakowski and Veblen 2003), stand susceptibility to beetle outbreak may be reduced until spruce regains dominance of that stand. Where land use has altered forest structures through changes in fire regimes and logging, the potential response to natural wind storms may have been altered.

The flat topography of the Grand Mesa may reduce susceptibility of forests to blowdown by reducing the acceleration of wind due to air pressure differences and wind turbulence, both of which increase with rugged topography. However, the high elevation of the mesa makes it likely that blowdowns do play a role in the development of higher elevation forests in the GM. In fact, in October 1997, 3,200 - 4,800 ha (8,000 - 12,000 ac) of spruce-fir forest were affected by a severe windstorm in the GM (US Forest Service *unpublished data*). In subsequent years, populations of spruce beetle increased in some stands affected by this blowdown and Forest personnel salvage logged infested stands to try to prevent a beetle outbreak (US Forest Service *unpublished data*).

5.2. Snow and Flood Events

Snow may cause disturbance to vegetation in several ways. Deep snows that persist late into a growing season may limit plant regeneration (Holway and Ward 1963, Butler et al. 1992). Furthermore, long lasting snow cover can lead to intensive overgrazing by ungulates in areas that are snowfree (Gilbert et al. 1970). Late snowmelt may also limit the success of trees by favoring the blackfelt snowmold (*Herpotrichia nigra*) on spruce and fir (Cooke 1955, Knight 1994). At low elevations in the Colorado Front Range, heavy snowstorms occasionally strip limbs and break the canopies of large conifers, or cause the collapse of dense patches of young conifers or aspens (Jack 1900).

Flooding causes local watershed changes that can significantly alter riparian ecosystems (Baker 1989, Malanson and Butler 1990). Riparian areas often contain highly diverse ecosystems, and geomorphic processes along stream channels both create and

remove habitats. Coarse woody debris is an especially important component for creating fish and aquatic habitats in stream channels (Harmon et al. 1986, Malanson and Butler 1990). Plant species diversity has been related to the area of sediment bars, stream sediment loads, and coarse woody debris washed into stream channels from upper reaches, much of which depends on watershed-scale vegetation patterns (Malanson and Butler 1990). However, vascular plant species richness at 115 riparian sites over a 300 km (190 mi; length of the western slope in Colorado did not spatially correlate with indicators of fluvial disturbance (Baker 1990b). Stand-origin dates of narrowleaf cottonwood (*Populus angustifolia*) indicate that changes in streamflows are important to the regeneration of this riparian tree (Baker 1990a).

Beaver were widespread in North America prior to Euro-American settlement. Early trapping is believed to have decimated many local populations of beaver in Colorado by the end of the 19th century (Wohl 2001), but the amount of beaver trapping, and thus possible beaver decline, in the GM is not known. Restoration efforts were initiated in Colorado to increase beaver populations for both ecological and economic reasons. Long-term trends in beaver populations in the GM are uncertain, but in 1947 the beaver population in Colorado was estimated to be 38,000 and stable (Denney 1952). The presence of beavers (*Castor canadensis*) has an influence on the riparian ecology of the Grand Mesa area. The increased availability of subsurface water associated with beaver dam construction may increase the growth of riparian vegetation (Wohl 2001). Furthermore, because beaver dams increase water storage along rivers, they contribute to more uniform stream flows. Conversely, removal of beavers may cause early failure of their dams, resulting in increased sediment transport and pronounced channel widening.

Many dams and reservoirs were constructed on the Grand Mesa early in the 20th century. Dams reduce the frequency and severity of flood events by regulating water flow. There has been no comprehensive study of the effects of fluvial disturbance on the vegetation of the GM, but because of the increase in regulation of water flow in the 20th century, changes in the dynamics of riparian vegetation are likely to have occurred. In fact, the *c.* 300 lakes and reservoirs on the GM that are diverted or dammed for irrigation and urban use has certainly altered the flow dynamics, and thereby the ecology of riparian habitats, in this region. Dams

change the thermal regimes and interrupt the movement of water, sediment, and biotic organisms in the rivers that they affect (Pepin et al. 2002) and it is probable that such changes have occurred in the waterways of the GM. There is a need to study the effects of dam and reservoir construction on the vegetation of the GM.

5.3. Fire

5.3.1. Fire Regimes Along the Montane Woodland-Subalpine Forest Gradient

The determinants of a fire regime, or the spatial and temporal occurrence of fire within a specified area, include fuel type and condition (which are also affected by fire history), ignition sources, topography, and weather at the time of ignition. Fuels are fundamentally controlled by productivity and decomposition rates permitted by the regional climate, but humans also can greatly modify fuel types through logging, fire suppression, road construction, or any other activity that alters fuel, stand, or landscape structure. The effects of such activities vary among ecosystems. For example, the effects of fire suppression will be more pronounced in systems that were previously limited by fuels due to frequent, low-severity fires.

Potential ignition sources include both humans and lightning. Both of these ignition sources have been widespread in the GM (Figure 5.1). Primarily due to climatic gradients associated with elevation, it is possible to generalize about fire regimes. However, there is much local variation associated with topography and vegetation structure that results in many exceptions to the general patterns described below. Most of these determinants of fire behavior are influenced by climate as well as human land use. For example, the frequency of lightning-caused fires has been greatest in the oak and pinyon-juniper cover types (Figure 5.2). In contrast, the spruce-fir cover type has the highest frequency of human-caused fires, probably due to greater recreational use of these forests. The pinyon-juniper cover type has the highest frequency of recorded large ($\geq$ 0.3 ac) fires between 1970 – 2001. While this corresponds to the largest proportion of area burned of any cover type, it still accounts for < 1% of the total extent of pinyon-juniper over 31 years.

Fire regimes differ among the different forest types (Table 5.1). Spatially and elevationally controlled

Figure 5.1a

20   0   20 Kilometers
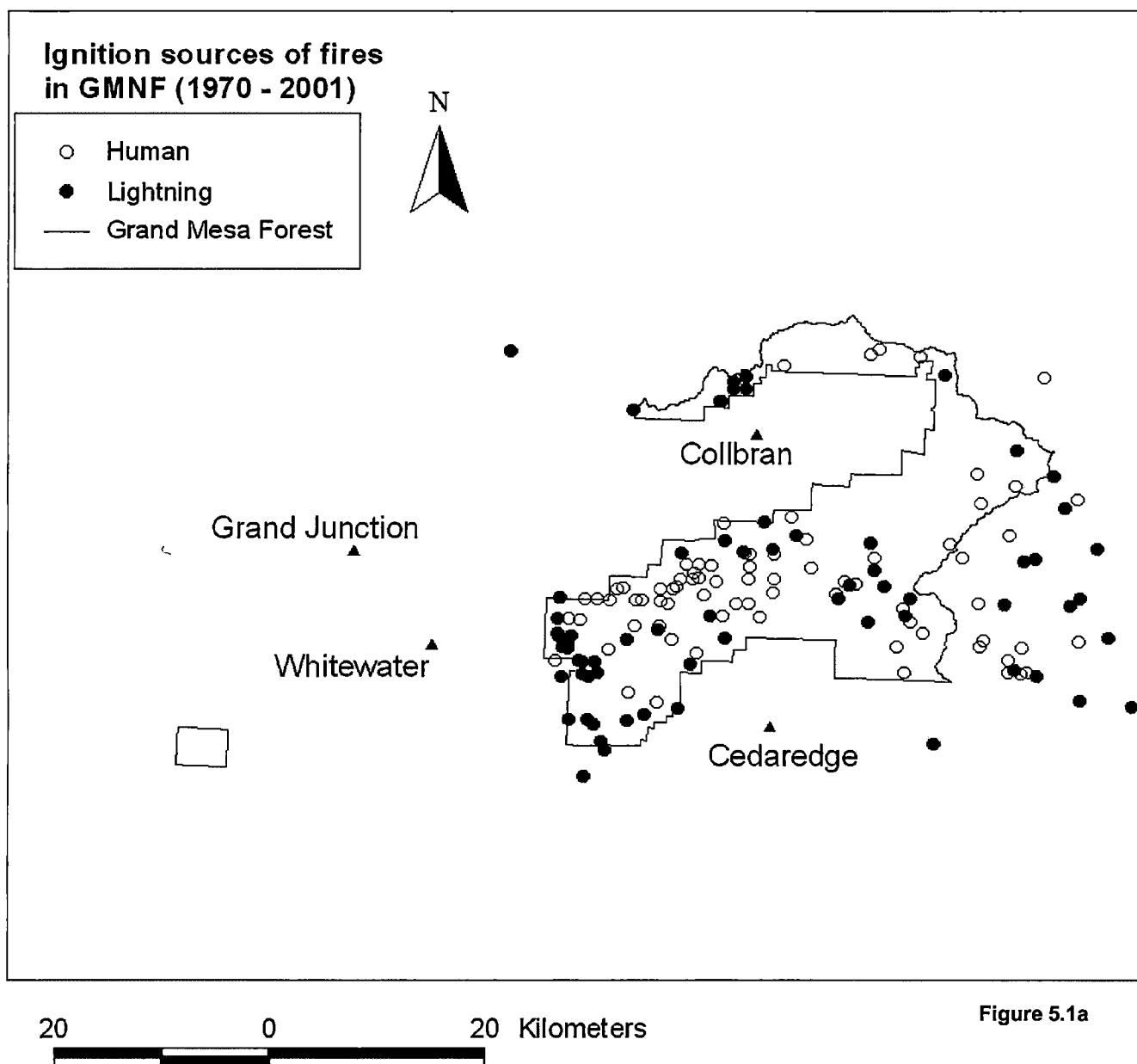
Figure 5.1—Map of (a) all fires and (b) large fires (≥ 0.3 acres) recorded from 1970 to 2001 in the GM.

forest types and understory are likely the primary reasons for differences in fire regimes (Alington 1998, Brown et al. 1999, 2000, Veblen et al. 2000, Kipfmueller and Baker 2000). Most fires in the relatively dry woodlands of pinyon-juniper and Gambel oak are smaller and relatively more frequent than in the more mesic subalpine zone. Fire frequency per area in pinyon-juniper woodlands has been higher than in any other cover type in the GM (Figure 5.2 a -

b). However, fires in this vegetation type are also often stand-replacing (Floyd *et al.* 2000, Eisenhart 2004, Floyd et al.2004). Fires in the nearby Gunnison Gorge area were historically stand replacing while lower severity fires have not had a significant influence on stand development (Eisenhart 2004). Fire frequency was higher in GBNP prior to Euro-American settlement and associated fire suppression and heavy livestock grazing. Similarly, in Mesa Verde National

Ignition sources of large (≥ 0.3 acre) fires in GMNF (1970 - 2001)

o Human
● Lightning
—— Grand Mesa Forest

N

Collbran

Grand Junction

Whitewater

Cedaredge

20   0   20 Kilometers
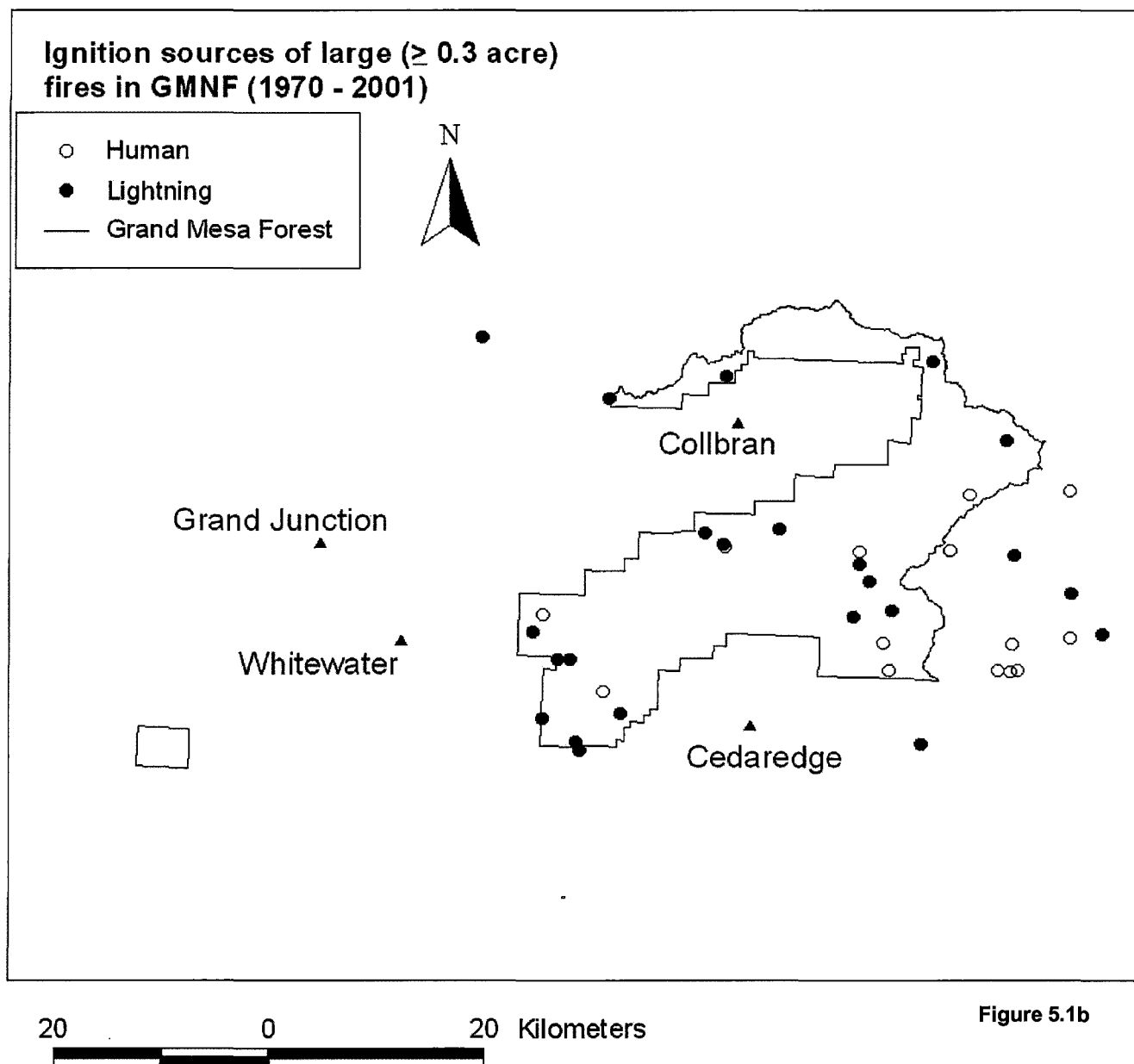
Figure 5.1b

Park (MVNP), pinyon-juniper woodlands had a historical fire turnover time of *c.* 400 years during the late 19th century (Floyd *et al.* 2000, 2004). There is no evidence that these stands were ever influenced by low-severity surface fires (Floyd et al.2004). Furthermore, it is unlikely that spreading, low-severity surface fires were ever common in any pinyon-juniper woodlands (Baker and Shinneman 2004). MVNP was affected by fire suppression in the early 20th century, which reduced fire occurrence below the frequency of the 19th century, to a turnover time of *c.* 1,200 years. However, during the latter half of the 20th century, this Park experienced an increase in fire occurrence that was associated primarily with favorable climatic conditions (as opposed to the effects of fire suppression; Floyd et al. 2004). This increased fire activity in the latter 20th century resulted in a fire turnover time of *c.* 400 years. Thus, the fire regime in MVNP in the late 20th century was similar to that of the 19th century.

Fire Frequency by Cover Type
(1970 - 2001)
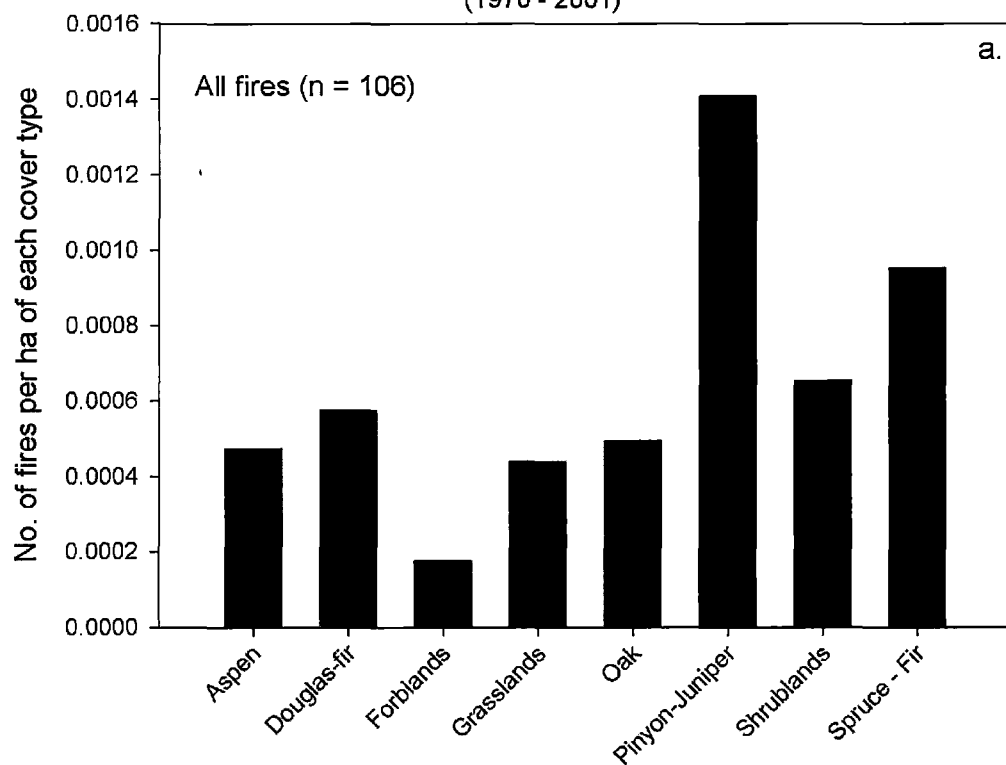
Figure 5.2—Number of (a) all fires, (b) large fires (≥ 0.3 acres), (c) lightning caused fires, and (d) human caused fires per hectare of each of eight cover types in Grand Mesa NF (1970-2001).
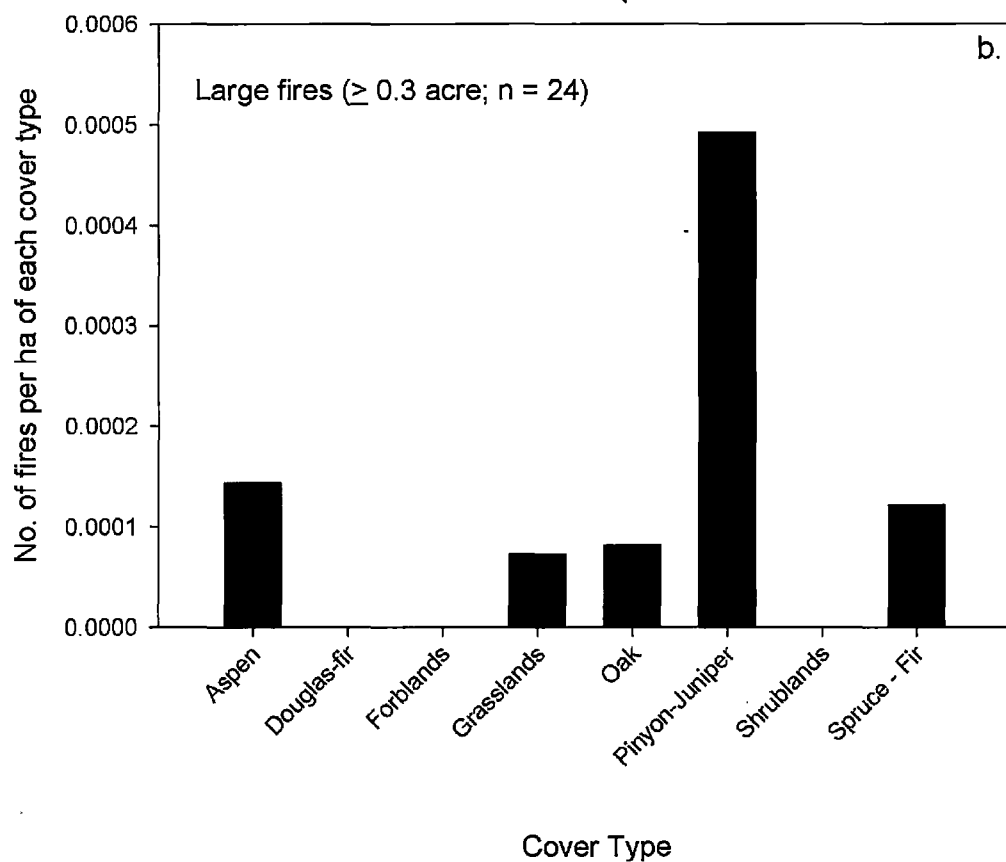
Cover Type

Fire Frequency by Cause
(1970 - 2001)

Lightning caused (n = 48)   c.

No. of fires per ha of each cover type

Cover types (x-axis): Aspen, Douglas-fir, Forblands, Grasslands, Oak, Pinyon-Juniper, Shrublands, Spruce - Fir

Human caused (n = 58)   d.

No. of fires per ha of each cover type

Cover types (x-axis): Aspen, Douglas-fir, Forblands, Grasslands, Oak, Pinyon-Juniper, Shrublands, Spruce - Fir
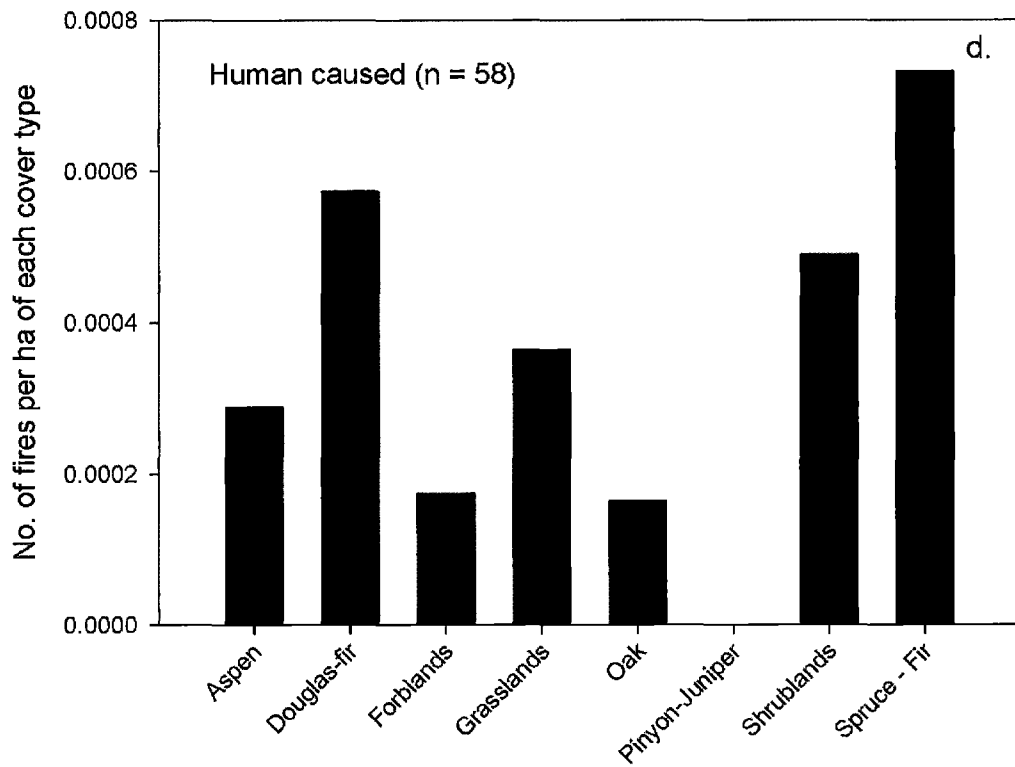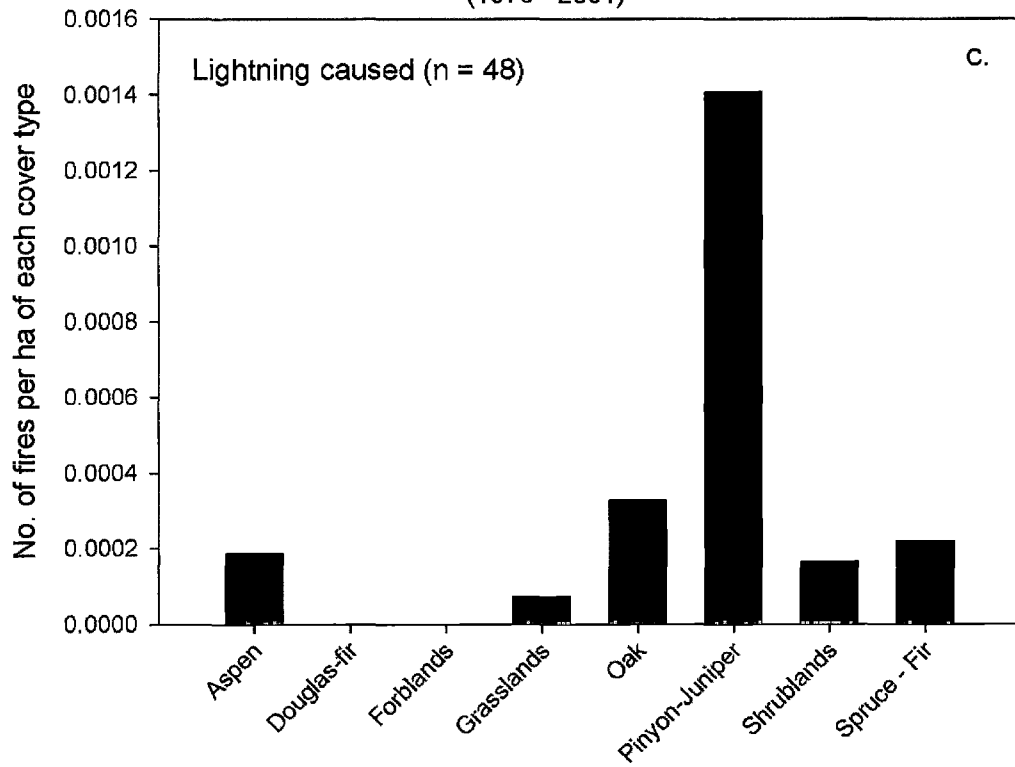
Cover Type

Table 5.1—Relative comparison of historic fire regimes in vegetation types of the GM.

Vegetation type	Dominant fire type	Relative fire frequency	Relative size
PJ and oak woodlands	Stand-replacing	Frequent	Less extensive
Aspen	Stand-replacing	Infrequent	More extensive
Spruce – fir	Stand-replacing	Infrequent	More extensive

Recent work in the neighboring Black Canyon of the Gunnison National Park suggests that some areas of woodland may burn very infrequently and that the fire regime of those woodlands is not outside of the HRV (Eisenhart 2004). It can be argued that, even during extreme drought, fires would have become larger in the absence of fire suppression efforts. While this argument is plausible, no data are available that document an ecological effect of fire suppression in these ecosystems. It is our interpretation that, while the behavior and effects of individual fires may have been altered by fire suppression activities, the net effect of fires has not been significantly altered. However, there may have been an increase in tree and shrub density in some lower elevation communities (Bradford 1998) but the cause for such an increase is not clear and is likely to be a complexity of fire, climate, and/or grazing. Although fire exclusion may have somewhat altered the relative abundance of young pinyon-juniper stands, there is no evidence that fires were formerly frequent (i.e., return intervals to same 1 ha of < 35 yrs); thus, fire exclusion has not transformed a surface fire regime to a stand-replacing fire regime. Based on existing studies in near-by woodlands, there is no convincing evidence that the fire regime of pinyon-juniper in GM has been fundamentally altered from the HRV. While studies in MVNP and elsewhere are useful by analogy, stand density and community type are likely to be different than in the GM area. Therefore, the results from such studies should be treated as hypothesis for the GM until they are tested.

In the higher elevation subalpine zone, the fire regime differs in a number of ways from that of montane woodlands. The primary differences between the fire regimes of the montane zone of PJ woodland and oak shrublands and that of the subalpine zone of spruce-fir and aspen forests are that (1) fires are more frequent in montane woodlands and (2) there is less continuous burn area in montane woodlands. There are several possible reasons for the differences in fire regimes between the montane woodlands and the subalpine forests. First of all, even though montane woodlands can have heavy fuel loads, the continuity of fuels is often low (Floyd et al. 2000), especially in comparison to the subalpine forests. Montane woodlands often have sparse fine fuels and many patches of rock or bare soil. Therefore, fires are likely to be contained in individual patches of fuel and do not spread over large areas, except under extreme weather conditions (Bruner and Klebenow 1979, Floyd et al. 2000). The second reason is that weather conditions that desiccate fuels sufficiently for fire spread are undoubtedly more common at lower elevations, occurring annually or almost annually.

The fire regime of aspen forests in the Southern Rockies is generally described as being one of relatively infrequent and stand-replacing fires (Veblen et al. 1994, Kulakowski and Veblen 2002, Kulakowski et al. 2003, Kulakowski et al. 2004, Romme et al. 2001). Although aspen can occur on xeric sites, many aspen stands are often very mesic and only burn under extremely dry conditions and may otherwise act as fire breaks. Because comparatively little fire history research has been conducted in this forest type, conclusions about fire regimes in aspen forests are tentative. Despite extensive searches for trees recording multiple fire scars in aspen areas, such evidence of repeated non-lethal fires has not been found. Thus, the current state of knowledge does not support the notion that non-lethal surface fires were formerly frequent and widespread in the aspen cover type Instead, aspen stands and surrounding spruce-fir forests had fire regimes characterized by infrequent stand-replacing fires. Such a fire regime is inherently variable at decadal and centennial time-scales. Thus, the reduced extent of burning in the aspen zone during the 20th century, compared to the late 19th century, is likely to be within a range of variability of the natural

fire regime over the past several centuries. Indeed, comparison of the extent of aspen on the GM in the late 19th century with its modern extent shows that aspen has increased in extent over the past century (Kulakowski et al. 2004).

Above the zone of aspen dominance, continuous canopy fuels of mesic, dense Engelmann spruce and subalpine fir forests, which often have developed fuel-ladders, permit widespread stand-replacing crown fires. Low decomposition rates in the subalpine zone may lead to great accumulation of fuels and, therefore, intensive crown fires. Extremely dry weather conditions conducive to fire spread are also not common in higher and more mesic areas, contributing to long intervals and great fuel buildup between fires (Clagg 1975, Romme and Knight 1981, Veblen 2000, Baker 2003, Sibold et al. 2006).

In spruce-fir forests of the subalpine zone in the southern Rockies, stand-replacing fires are well documented as having the greatest impacts on forest structure. In areas of continuous forest in the subalpine zone, vast areas have burned in single stand-replacing events as indicated by extensive even-aged tree populations (Sudworth 1900, Whipple and Dix 1979, Romme and Knight 1981, Veblen 1986a, Aplet et al. 1988, Parker and Parker 1994). In northern RMNP, fire history in a small watershed has been analyzed by radiocarbon dating of thick charcoal bands in a sediment core covering the last 6250 years; the results imply that most or all of this small (24 km^2) subalpine watershed burns on average once every 480 years (Madole 1997). This is consistent with fire rotation periods (time period to disturb the entire area once by fire) estimated from tree-ring dating of stand-replacing fires of 182 years and 308 years for study areas of several thousand hectares of subalpine forests in the Medicine Bow Range of southeastern Wyoming and in southeastern RMNP, respectively (Kipfmueller and Baker 2000, Sibold 2001, Buechling and Baker 2004, Sibold et al.2006). Similarly, long intervals have also been reported in western Colorado. For example, fire turnover time in a small (c. 35 km-sq.) subalpine watershed in White River National Forest was reported to be 521 years (Veblen *et al.* 1994). Paleoecological evidence also suggests that stand-replacing fires with long (≥ centennial) return intervals have been important in the subalpine forests of western Colorado for much of the Holocene (Fall 1997).

The extensive even-aged structures of spruce-fir forests in western Colorado and elsewhere in the southern Rockies indicate that these forests are shaped primarily by infrequent (FRI of 100 to > 300 years) but lethal stand-replacing fires (Veblen 1986a, Clagg 1975, Romme 1982, Romme and Knight 1981, Peet 2000, Veblen et al. 1994, Kipfmueller and Baker 2000, Kulakowski and Veblen 2002, Kulakowski et al. 2003). However, these fire regimes are spatially variable within the subalpine zone. In mesic habitats of dense spruce-fir forests in the subalpine zone, fire regimes are dominated by infrequent stand-replacing fires. On the other hand, in some spruce-fir and lodgepole pine stands in northern Colorado and southern Wyoming, surface fires that are not lethal to canopy trees can also occur even though the structure of the forest is shaped primarily by stand-replacing fires (Kipfmueller and Baker 2000, Sibold 2001, Kulakowski and Veblen 2002, Kulakowski et al. 2003, Sibold et al.2006). Limited fire-scar data from Colorado indicate that frequent, extensive surface fires were not the norm in subalpine forests. Instead, the surface fires documented in these forests were infrequent (typically only one widespread surface fire event in stands < 150 years old) and of limited extent (Sibold 2001, Kulakowski et al. 2003, Sibold et al.2006). A systematic search for fire-scarred trees during the summer of 2002 failed to yield many trees with multiple fire scars (Kulakowski and Veblen *unpublished data*) which implies that surface fires were not important in this area. Over a sampled area of *c.* 2000 km-sq. in the GM, the presence of a relatively small number of post-fire cohorts originating since c. 1700 (Kulakowski and Veblen in press) suggests that the GM fits this general pattern of infrequent but widespread fires in spruce-fir forests. Although fire suppression may have prevented some small fires from becoming widespread, the scarcity of widespread fires in the 20th century is not atypical in comparison with previous centuries. The long natural fire return intervals of several hundred years in mesic spruce-fir forests (Romme and Knight 1981, Veblen et al. 1994, Sibold et al. 2006) would make it unlikely that a significant change has occurred in fire frequency during the relatively short period (c. 80 years) of fire suppression.

Historical photographs show the occurrence of widespread stand-replacing fires across the GM (Sudworth 1900; Plates 1 - 2). These fires were reported to have occurred primarily in 1878 - 1879 and less extensive fires also occurred in 1883 - 1885 and 1890 - 1892 (Sudworth 1900). Dendroecological data

41

Plate 1—View on slope of Mesa Creek looking west along north side to end of north prong of Grand Mesa, showing extensive burn. Photo: Sudworth, Oct. 22, 1898 (#365).

suggest that fire around the 1850s also affected forests of the GM (Kulakowski and Veblen *unpublished data*). These data also show that the vast majority of spruce – fir stands in the Grand Mesa area are dominated by distinct cohorts that most likely established following stand-replacing fires. However, the presence of many remnant trees that survived otherwise stand-replacing fires suggests a slightly lower intensity of fires in this area in comparison to other subalpine forests in Colorado (e.g. Veblen et al. 1994, Kulakowski and Veblen 2002). Historical photographs and accounts are consistent with this interpretation (Sudworth 1900; Plate 3). Sudworth (1900) wrote:

... the most extensive burned areas have their surviving groups or scattered single trees. Some are near the borders of these great burns and others are scattered over the greater part or whole of the burned areas.

The spruce-fir forests of GM National Forest occur primarily on top of the mesa in areas of gentle topography. This gentle topography may reduce the severity of fires in these forests in comparison to similar forests on steeper terrain. The basalt cap of the mesa and the level topography also contribute to poor drainage, and favor the formation of meadows, which may act as fire breaks. In addition to these topographic

Plate 2—View looking west by north up along the north side prong of Thompson Creek, showing long bare ridge on north side and extensive burn in foreground. Photo: Sudworth, Sept. 28, 1898 (223).

and geologic factors which favor meadows, some meadows may originate when unfavorable climate impedes tree regeneration after forest burning. Conversely, favorable climatic conditions could facilitate tree invasion into meadows.

In general, the subalpine forests of Grand Mesa appear to fit the regional pattern of stand-replacing fires in the spruce-fir cover type. However, the relatively large area of meadows on the Grand Mesa probably has affected fire spread and intensity by serving as fire breaks. Thus, although fire history in subalpine forests needs further study, generally in Colorado and specifically in Grand Mesa, the data

presently available do not suggest that subalpine forests of the GM were characterized by frequent and extensive surface fires. Instead of current stand structures showing any influence of reduced fire frequency, they primarily show the effects of stand-replacing fires that were particularly extensive in the second half of the 19th century.

At a broad spatial scale, the landscape patterns generated by fires differ between spruce-fir, aspen, pinyon-juniper, and oak vegetation types (Table 5.1). Widespread, infrequent stand-replacing fires in spruce-fir forest types appear to result in relatively larger patches of even-aged forests. In contrast, the patches
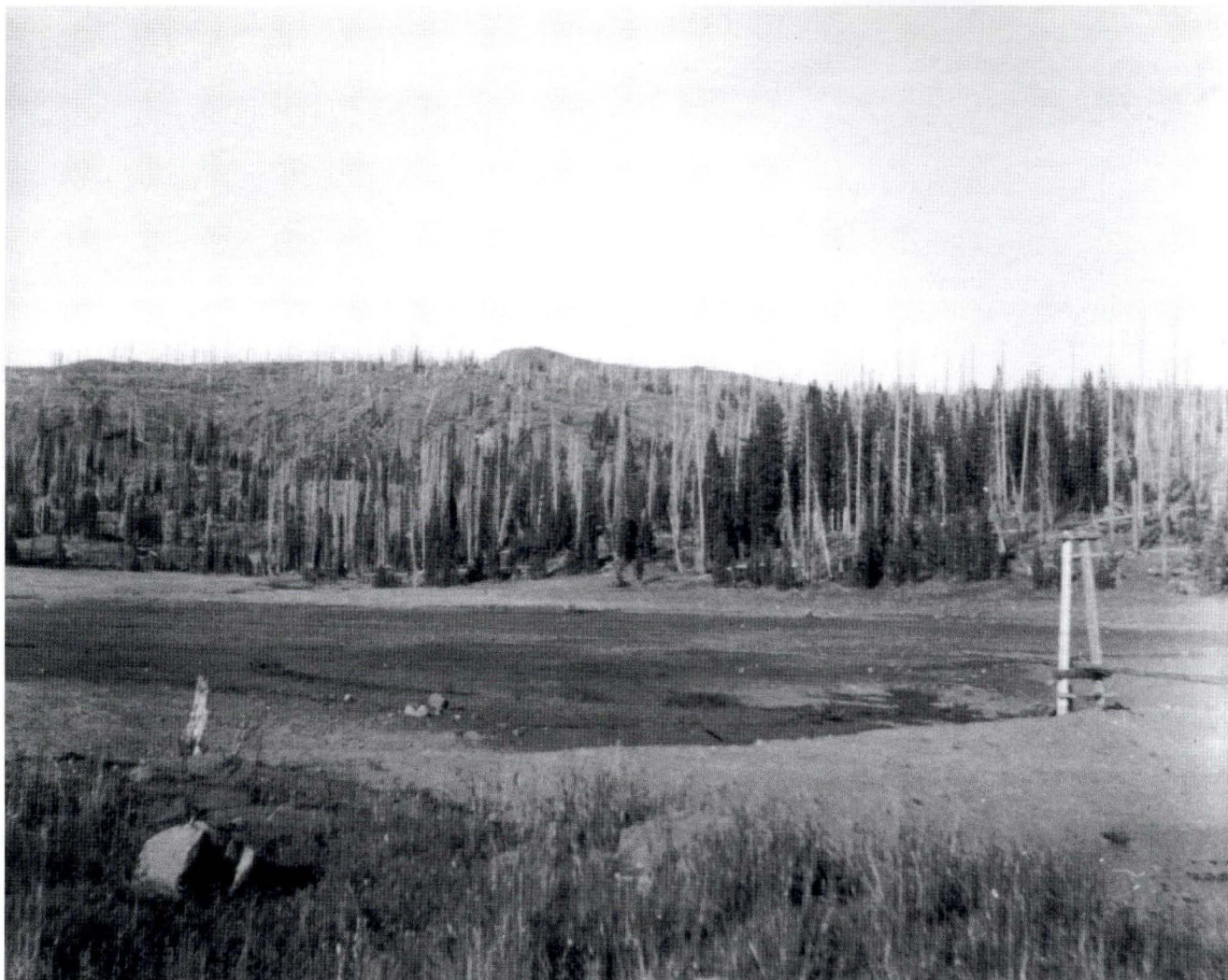
Plate 3.—View from the south side of reservoir on head of Leroux Creek looking north over the slope showing extensive fire with patches of remnant trees. Photo: Sudworth, Oct. 7, 1898 (297).

that result from the more frequent but smaller stand-replacing fires in pinyon-juniper and Gambel oak woodlands create more fine-scale heterogeneity in stand age and structure. Aspen forests extend over an elevational zone that is between spruce-fir forests and PJ and oak woodlands. In the GM, aspen forests appear to be dominated by a regime of relatively infrequent and large fires. However it is likely that variation exists such that infrequent fires may be more dominant at higher elevations and by smaller more frequent fires may be more important at the lowest elevations of aspen's extent.

5.3.2. Influences of Inter-Annual Climatic Variation on Fire Regimes

The relationship of fire occurrence and behavior to fire weather at temporal scales of hours to seasons is well understood and used to predict short-term fire hazard (Rothermel 1983). At an inter-annual scale, synchronous occurrence of fire-scar dates from areas too large for fire to have spread from a single ignition point is strong evidence that regional climate is influencing fire regimes. For the area from southern Wyoming to southern Colorado, widespread burning in the 1870s to 1880s was recorded in early, albeit fragmentary, documentary sources (Sudworth 1900, Plummer 1912, Ingwall 1923), as well as in tree-ring studies of fire history (i.e., Skinner and Laven 1983,

Zimmerman and Laven 1984, Goldblum and Veblen 1992, Kipfmueller 1997, Veblen et al. 2000, Brown et al. 1999, Kulakowski et al. 2003). Individual years that recorded fire scars at disjunct locations over this large area include 1654, 1684, 1809, 1813, 1842, 1851,1859-1860, 1871-1872, 1879-1880, and 1893-94 (Kipfmueller 1997, Alington 1998, Brown et al. 1999, Veblen et al. 2000, Donnegan et al. 2001, Sherriff et al. 2001, Brown and Carpenter 2001, Sibold 2001, Kulakowski and Veblen 2002, Sibold et al. 2006). Such synchrony of fire years suggests that at a regional scale extreme weather increases fire hazard over extensive areas from southern Wyoming to southern Colorado. Indeed, tree rings sampled at numerous sites in northern Colorado (Cook et al. 1998, Veblen et al. unpublished data) indicates that all of the major fire years listed above correspond with significant drought during the year of the fire and/or the year immediately preceding the fire year. Similarly, for Grand Mesa, Uncompahgre, and Gunnison National Forests the Forest Service records of fire extent and local climate records indicate that area burned is strongly associated with drought. While the number of ignitions was variable, and most likely not limiting, relatively few years experienced widespread (> 200 hectares per year) burning (Figure 5.3). Such marked variation in the area burned per year with only 6 years recording > 200 hectares burned in a year is strongly suggestive that fire spread is dependent on infrequently occurring weather conditions. While the small number of fires does not permit statistical analysis of the effect of weather on fire, these years of widespread burning tended to occur during years with below average spring or summer precipitation.

In the case of the montane zone of MVNP in southwestern Colorado characterized by PJ woodlands and oak shrublands, years of widespread fire have also been shown to be highly dependent on unusual drought (Floyd et al. 2000). Conversely, prolonged periods of increased moisture may reduce the number of large fires. All large (> 10 acres) fires in MVNP from 1926-1977 occurred in late June – early July, except for one which occurred in August and all occurred under conditions of high wind and prolonged drought (Floyd et al. 2000). However, the complete effects of climatic variability on the fire regime of the PJ woodland of MVNP are not completely understood. For Pinyon-Juniper and Oak shrublands in the GM it is not clear whether fire occurrence is primarily associated with single years of below average precipitation, periods of prolonged (e.g. decadal) drought, or periods of increased climatic variability (which may increase the production and desiccation of fine fuels).

5.3.3. Anthropogenic Influences on Fire Regimes

Extensive burning during the 19th Century: Widespread fire occurrence during the second half of the 19th century has been documented for many areas in the southern Rocky Mountains (Rowdabaugh 1978, Laven et al. 1980, Skinner and Laven 1983, Zimmerman and Laven 1984, Goldblum and Veblen 1992, Kipfmueller 1997, Donnegan 2000, Veblen et al. 2000, Brown and Carpenter 2001). While a large area of subalpine forest did burn in 1879 and the 1880s in western Colorado (Sudworth 1900, Kulakowski and Veblen 2002, Kulakowski et al. 2003, Kulakowski and Veblen unpublished data, Plate 1 - 3), the occurrence of extensive fires during these very dry years is not outside of the HRV for this forest type. However, because large fires in subalpine forests are infrequent, and dendrochronological reconstructions of fire regimes are limited by the longevity of the tree species (usually < 700 years) and generally rapid decomposition rates in mesic subalpine forests, it is difficult to establish long-term trends in the history of stand-replacing fires in this forest type.

The latter half of the 19th century was a time of severe droughts and high inter-annual climatic variability favorable to extensive fires in the Colorado Rockies (Veblen et al. 2000, Donnegan et al. 2001) but it was also a time of increased anthropogenic ignitions. Across much of Colorado's eastern slope, intentional or accidental ignitions by Euro-American settlers were frequent (Veblen and Lorenz 1986). During the Euro-American settlement period (c. 1850 to 1910) in Colorado, fires were frequently set by Euro-Americans to facilitate prospecting, to justify salvage logging, or to clear brush for ranching (Tice 1872, Fossett 1880, Jack 1900, Sudworth 1900, Wier 1987). It also appears that the influence of Euro-American ignition, especially in association with near-by coal mining, contributed to burning in the area of the GM (Sudworth 1900). Although there was a relatively small presence of Euro-Americans around the GM in the latter half of the 19th century, their influences through burning and land use were likely to have been important in shaping the forest landscape. Whereas in other parts of Colorado, Native Americans were blamed for

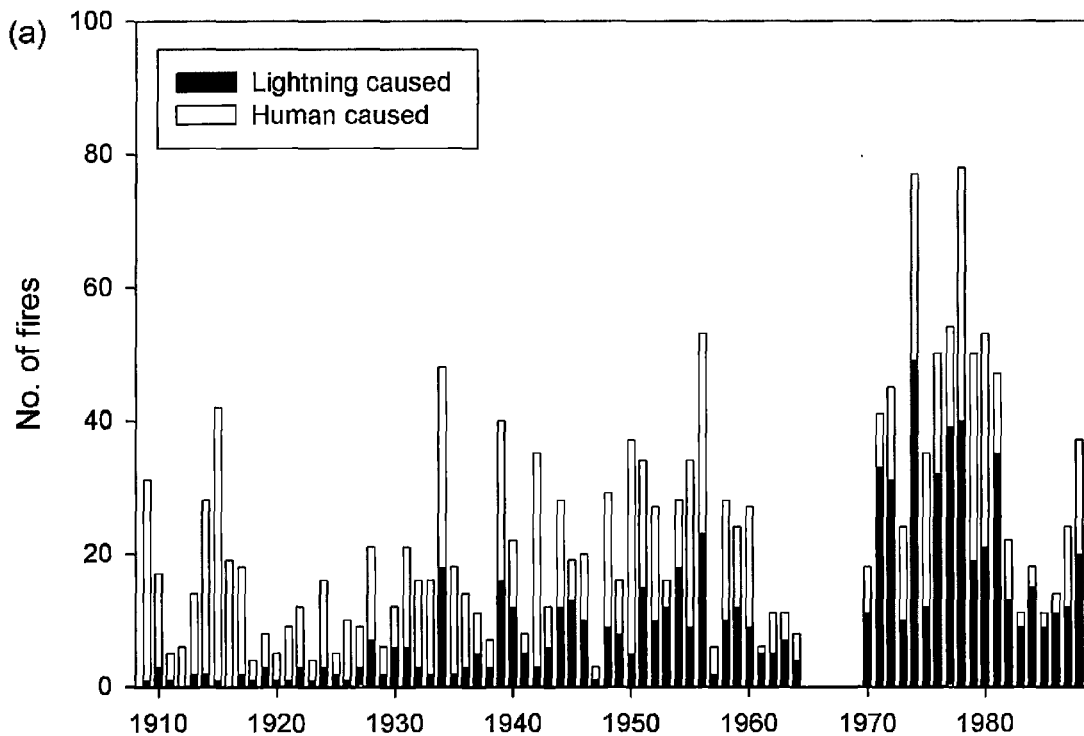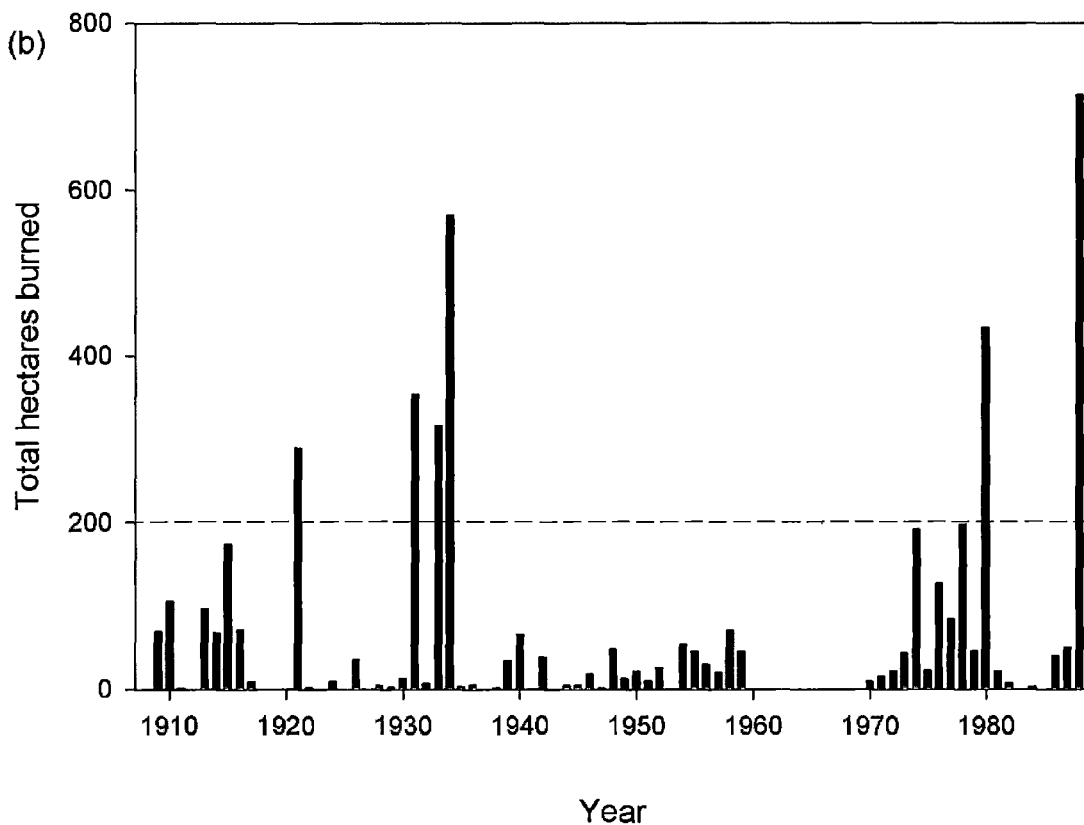## Fires in GMUG Forests
(1909 - 1988; n = 1743)

Figure 5.3—Number (a) of lightning and human caused fires and (b) total hectares burned per year in Grand Mesa, Uncompahgre, and Gunnison National Forests (1909 - 1988). Dashed line indicates threshold of 200 hectares burned per year. Source: USFS.

intentionally and extensively burning forests, both white (Sudworth 1900) and Native accounts (US House of Representatives 1880) attribute the spread of large fires in the GM in the late 19th century to exceptionally dry years regardless of whether their origins were natural or anthropogenic (See Section 3.4). The weather was so favorable for fire occurrence and spread in the late 19th century that either natural lightning ignition or accidental negligence by whites or Utes would have been likely to start extensive blazes. As described in section 5.3.2 the mid- and late-19th century was a period that was climatically conducive to widespread fire in the Colorado Rockies. This was a time of both severe drought as well as increased year-to-year variation in moisture availability. Although it is difficult to exactly determine the roles played by climate versus humans, the historical accounts strongly imply that climate was the most significant factor in fire occurrence in the second half of the 19th century in the area of GM.

Reduced fire occurrence during the 20th Century: The effects of fire suppression policy during the 20th century vary among forest types. A pattern of reduced fire frequency during the 20th century has been consistently found in studies of montane and upper montane fire history from the northern Front Range to the Sangre de Cristo Range (Rowdabaugh 1978, Laven et al. 1980, Skinner and Laven 1983, Goldblum and Veblen 1992, Alington 1998, Brown et al. 1999, Veblen et al. 2000). However, some pinyon-juniper and oak woodlands experienced a reduced fire frequency during the early 20th century, and a higher fire frequency, that was similar to HRV, during the latter half of the 20th century (Floyd et al. 2000). In comparison to the subalpine zone, the historic fire regime of the montane zone consisted of more frequent fires. Thus, the historic fire regime of the montane zone is inherently more susceptible to significant modification by fire suppression. The modern fire exclusion period beginning in the early 1900s refers to both suppression of lightning-ignited fires and cessation of widespread, intentional burning by humans. Fuel reductions due to heavy grazing in the late 19th and early 20th centuries also may have contributed to the decline in fire frequency in some ecosystems near the turn of the century, which in many studies pre-dates effective fire-suppression technology by one or several decades (Veblen 2000).

Fire suppression efforts in National Forests have been substantial and most recent fires have been small on Colorado's western slope, with the notable exception of fires in 2002. However, in the context of HRV, the actual impact of fire exclusion is uncertain in both montane and subalpine forests of Colorado's west slope. Indeed, for the main subalpine zone of spruce-fir and lodgepole pine forests of Yellowstone and Rocky Mountain National Parks, Romme and Despain (1989) and Clagg (1975) questioned if fire suppression has significantly altered the natural fire regimes. In contrast, in the Medicine Bow Range of southern Wyoming, Kipfmueller and Baker (2000) describe a substantial decrease in the area burned by stand-replacing fire after the beginning of fire suppression in c. 1912. In the Wild Basin area of RMNP, the area burned in the 20th century also is less than that burned in the 19th century; however, here the total area burned in the 20th century is actually greater than the area burned during the 18th century (Sibold 2001) demonstrating the high degree of natural variability in area burned in the subalpine zone. Because of this high variability in areas burned at a centennial scale and the long fire intervals typical of subalpine forests, it is not certain that fire suppression has significantly reduced area burned over its long-term average in the subalpine zone. Particularly for the wetter spruce-fir habitats, it is difficult to assess the long-term effectiveness of fire suppression where large fire events may be limited to years of exceptional weather conditions occurring a few times per century. While the fire-free period of the 20th century is not outside of the HRV, it is possible that in some subalpine forests, suppression efforts have prevented some small fires from becoming extensive fires. Thus, the area of young (< 80 years old) subalpine forest may be smaller than what it would have been in the absence of fire suppression. However, the magnitude of this suspected trend is believed to be small.

Many resource managers believe fire suppression in the GM has effectively reduced the area burned in spruce-fir forests in the 20th century. In the absence of a century-long controlled experiment in which one area is subject to fire suppression and another is not, it is difficult or impossible to test that belief. Although it is not possible to quantitatively assess the magnitude of any reduction in area burned due to fire suppression activities, it is known that many large areas of spruce-fir forests in the Colorado Rockies experienced century-long periods of fire absence similar to the relatively fire-free period of the 20th century. Given

the long intervals between years of widespread severe fires in the spruce-fir cover type, the modern fire regimes of these forests are not greatly, if at all, outside of their historic range of variability. Natural, long fire-free intervals commonly occurred in large areas of spruce-fir forests in the southern Rockies (Sibold 2001, Sibold et al.2006, Kulakowski and Veblen 2002), and are reason for challenging the notion that modern fire exclusion has resulted in unnatural fuel accumulations or forest health conditions. Furthermore, because widespread surface fires did not play a significant role in these subalpine forests, fire suppression is unlikely to have significantly altered this component of the fire regime at the stand scale. Instead, it is more likely that weather exerts the primary control over stand-replacement fire regimes in subalpine forests.

For the zone of montane woodlands, it is less certain if the modern fire regime is within the HRV. For example, for pinyon woodlands in Great Basin N.P., Gruell (1999) concluded that modern fire frequencies are lower than the historic average. In contrast, Floyd et al. (2000) describe a modern fire regime that is similar to the historic pattern. Working nearby the GM on the Uncompaghre Plateau and in the Gunnison Gorge and Black Canyon of Gunnison National Park, Eisenhart (2004) concluded that the historic fire regime consisted of severe fires occurring at long intervals so that 20th century fire exclusion would have had little if any impact on the historic fire regime. Nevertheless, resource managers have observed areas of PJ woodland where lightning-ignited fires were frequent occurrences and believe that suppression activities have prevented some of these fires from becoming widespread events. Unfortunately, data (either written records or tree-ring records) are not available for the PJ woodlands of GM to allow a quantitative comparison of pre- and post-1900 fire regimes. It seems likely that fire suppression activities would have reduced the spread of some or many fire events in the 20th century in this cover type, but the magnitude of any overall reduction in fire extent is impossible to measure. What is clearly emerging from recent research in PJ woodlands in Colorado (Floyd et al. 2000, 2003, Eisenhart 2004) is that there is no evidence for the widespread application of a model of an historic fire regime of formerly frequent surface fires in PJ woodlands. Instead, these woodlands have been shaped primarily by severe, stand-replacing fires occurring at long intervals. This implies that even if fire suppression in the 20th century

significantly reduced the area burned in these woodlands, it has not resulted in a shift in fire type from low intensity surface fires to stand-replacing fires. In other words, the fire suppression and fuels accumulation model used for explaining the shift in fire behavior from historic surface fires to modern crown fires in many areas of ponderosa pine forests in Arizona (Allen et al. 2002) does not apply to PJ woodlands in southwestern Colorado. Specifically for the PJ woodlands of GM, despite some uncertainty and the possibility of local exceptions, we tentatively conclude that fire suppression has not moved the fire regime greatly, if at all, outside the HRV.

5.4. Insects and Diseases

A number of insects and pathogens are ecologically important in the forests of western Colorado. All of these species are natives of western North America and have co-evolved with their hosts for millennia, thus, in some capacity, their effect on these forests has been a part of HRV. However, in most cases, the historical extent and magnitude of these effects are not known.

In the forests of Colorado's western slope the important forest insect pests include Douglas-fir beetle (*D. pseudotsugae* Hopkins), spruce beetle (*D. rufipennis* Kirby), western tent caterpillar (*Malacosoma californicum* Packard), pandora moth (*Coloradia pandora* Blake), Douglas-fir tussock moth (*Orgyia pseudotsugata* McDunnough), western spruce budworm (*Choristoneura occidentalis* Freeman), and the pinyon ips (*Ips confusus*) (Schmid and Mata 1996, Leatherman and Kondratieff 2003). Important pathogens affecting many of the conifers include dwarf mistletoe (*Arceuthobium* spp.) and *Armillaria* root diseases. Each of these disturbance factors and how their occurrence potentially could have been modified by humans will be considered separately below. Other decay, root, and foliage diseases and pathogens affect Colorado forests but are not included, either because they are minor or because no potential linkage to human activities could be argued.

It is important to note that even for contemporary ecosystems, quantitative data on populations (e.g., numbers of insects per tree or area) are rare. An insect population may be qualitatively classified as at endemic or epidemic levels, or in terms of the amount of tree mortality they cause over an area (Schmid and Mata 1996). It is often difficult or arbitrary to define the temporal limits of insect outbreaks which are

typically spatially heterogeneous within a particular landscape or region.

5.4.1. Bark Beetles

All of the bark beetles tend to attack larger trees (typically > 10 to 20 cm diameter; > 4 to 8 inches diameter), and their attacks are normally lethal (Schmid and Mata 1996). They bore through the bark, create egg galleries, mate, and deposit eggs in the phloem layer. They carry with them fungi, which in conjunction with the beetle's excavations, results in blockage of water- and nutrient-conducting tissues, thus killing the tree.

Spruce beetle: The spruce beetle (SB) in the southern Rocky Mountains mainly infests Engelmann spruce, but can also attack large lodgepole pine, if the pine is present (Alexander 1987, Schmid and Mata 1996). Endemic SB populations infest fallen trees and scattered live trees but beetle populations can reach epidemic levels and cause outbreaks that kill most canopy spruce over extensive areas. Epidemic populations are thus high and cause readily noticed or significant amounts of tree mortality (Schmid and Mata 1996). Spruce < 10 cm (4 inches) in diameter usually are not attacked, nor is subalpine fir, and their accelerated growth following the death of surrounding canopy trees can be used to date outbreaks (Veblen et al. 1991b). Spruce beetles preferentially attack larger trees and trees in stands with structural attributes associated with advanced stages of development (e.g. mean dbh > 40 cm, basal area > 34 m-sq per ha, and more than 65% spruce in the canopy) and stands in valley bottoms (Schmid and Hinds 1974, Schmid and Frye 1976).

SB outbreaks result in a massive shift in dominance in basal area from spruce to fir. This shift is due both to mortality of large spruce and in-growth of formerly suppressed seedlings and saplings of subalpine fir that are typically the most abundant tree species in the understory (Veblen et al. 1991c; Plate 5). Some new seedling establishment of spruce and fir may also occur. SB outbreaks may increase the hazard of severe fire during a relatively short period of 2 to 5 years when fine fuels from dead needles and twigs are more abundant. While the slow decay and fall rate of the dead-standing trees (Hinds et al. 1965) could imply that there may be an increased potential for more fire over many decades, empirical data from nearby White River National Forest does not support this notion(Bebi et al. 2003). Following a severe 1940s beetle outbreak, there was no increase in fire occurrence in beetle affected stands during the 50 years following the outbreak, nor were there extensive and severe fires (Bebi et al. 2003). Likewise, analysis of surface fire spread has not demonstrated an increase in fire spread in beetle-affected stands (Kulakowski et al. 2003). These studies suggest that any increase in fire hazard that follows beetle outbreak may have to coincide with appropriate climatic conditions if it is to be realized. Otherwise, the fire regime of subalpine spruce-fir forests is more strongly controlled by extremely dry weather than by local differences in fuel conditions, such as those following beetle outbreak.

Interactions among natural disturbances are also important in shaping Colorado subalpine forests and may be a predictable component of the disturbance regime of these forests. For example, post-fire stands a re reduced in their susceptibility to beetle outbreak for at least c. 70 years following stand initiation (Veblen et al.1994, Kulakowski et al. 2003). However, a broader-scale analysis of fire – beetle interactions suggests that, while such interactions may be widespread, they may also vary spatially in relationship to topographic variables (Bebi et al. 2003). Blowdowns or accumulation of logging debris are usually the immediate triggers of outbreaks (Schmid and Frye 1977), which is an important distinction from outbreaks of mountain pine or Douglas-fir beetle. Fallen trees provide abundant food and winter protection for promoting the growth of SB populations which then attack living trees. A strong windstorm in 1939 in northwestern Colorado is believed responsible for the largest recorded epidemic of the 20th century in the Rocky Mountain region (Massey and Wygant 1954). However, blowdowns do not always trigger spruce beetle outbreaks, even in highly susceptible forest types. For example, in Routt National Forest, there was no increase in beetle-caused mortality following a 1934 blowdown (Kulakowski and Veblen 2003). This suggests that there may be regional variations in forest susceptibility to beetle outbreaks and therefore in the interactions between blowdowns and beetle outbreaks. However, an outbreak has initiated following the severe 1997 blowdown that affected Routt National Forest (USDA Forest Service 2002 b). Beetle populations increased in the few years immediately following the blowdown and recently, these populations began attacking live trees in Routt

Plate 5—View of the existing stunted Engelmann spruce and subalpine fir timber on the north side of the slide-hill. All the large timber dead from some natural cause. Mortality pattern of large spruce, but not of smaller spruce and subalpine fir is typical of beetle outbreak. Photo: Sudworth, Oct. 2, 1898 (257).

and Jackson Counties. Beetle-killed trees have also been reported in the Flat Tops area in 2001. As of February 2004, it is impossible to judge what the likely extent or severity of this outbreak will be.

At least two regional SB outbreaks have affected the GM area in the past two centuries. The 1940s SB outbreak killed 4.3 billion board feet of timber in White River, Grand Mesa, and Routt National Forests (Massey and Wygant 1954). This outbreak was less severe in the Grand Mesa than in more northern areas, such as White River National Forest (Schmid and

Hinds 1974). Some forests of the GM were greatly affected by this outbreak, while other stands were less severely affected or unaffected (Figure 5.4; Veblen et al. 1991c). In affected stands, the outbreak resulted in a coincidence of mortality of large spruce and a release of the previously suppressed subalpine fir and small diameter spruce, thus favoring the fir.

Tree-ring methods and historical photographs document the occurrence of an earlier regional-scale SB outbreak in the mid-1800s in western Colorado that likely was at least as extensive as the 1940s outbreak
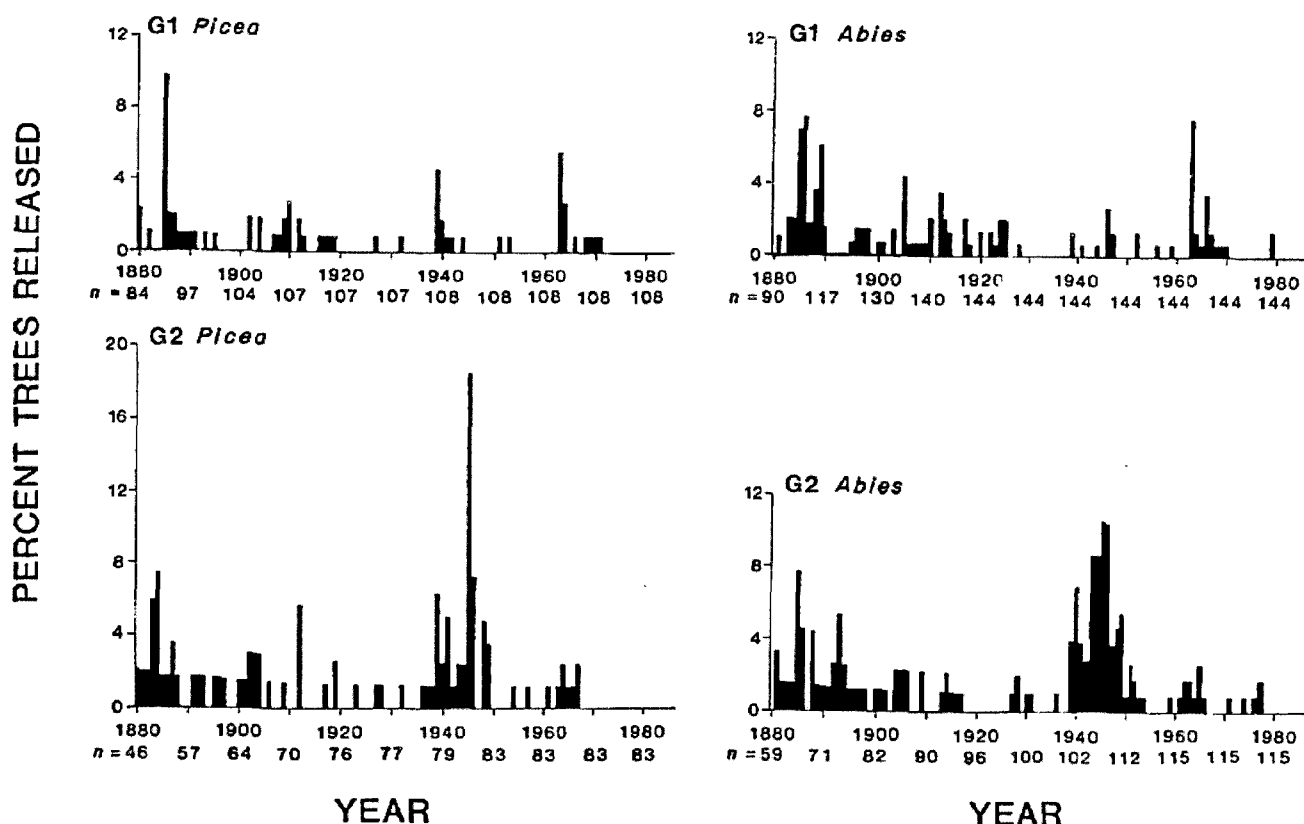
Figure 5.4—Percentage of live trees released show effect of spruce beetle outbreak in the 1940s. A release is an increase in mean ring width of > 250% when adjacent groups of five rings are compared. The number of surviving trees alive at the beginning of each decade is given by *n*. G1 is Cottonwood Lake and G2 is Big Creek Reservoir. Figure from Veblen *et al.* 1991c.

(Baker and Veblen 1990, Veblen et al. 1991b, 1994, Eisenhart and Veblen 2000; Plate 4 - 6). This outbreak also affected parts of the GM National Forest in *c.* 1850 (Figure 5.5; Veblen *et al.* 1991 c; Kulakowski and Veblen *in press*). The occurrence of this widespread outbreak, as well as the occurrence of outbreaks recorded in fossil records (Feiler and Anderson 1993), was prior to any significant impact of Euro-Americans on the subalpine forests of northwestern Colorado in the form of either logging or fire suppression. Therefore, severe and widespread SB outbreaks are clearly a natural component of disturbance regimes in the subalpine zone.

The frequency of severe SB outbreaks in the same stand is usually limited by lack of trees large enough to be susceptible to beetle attack (Schmid and Frye 1977). At a stand scale, lack of large-diameter spruce for 70 to 100 years after a severe outbreak or a stand-replacing

fire prevents that stand from being attacked even when surrounding older forest is infested (Veblen et al. 1994, Schmid and Mata 1996). However, at a regional scale, western Colorado was affected by two major outbreaks in a span of only c.100 years (Veblen et al. 1991b). At a smaller scale in a 594 ha (1467 ac) area of subalpine forest in northwestern Colorado, tree-ring methods documented three extensive SB outbreaks since the early 1700s (Veblen et al. 1994). Mean return interval and rotation period for SB outbreaks were 117 and 259 years, respectively, which made disturbance by SB more important, at least spatio-temporally, than disturbance by fire in this valley. In the GM, outbreaks appear to have been less severe, allowing some stands to be affected by both the mid-1800s outbreak and the 1940s outbreak (Kulakowski and Veblen *unpublished data*).

Because logging operations can trigger SB

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Plate 4—View of head of Beaver Creek, prong of Surface Creek, showing east side of canyon with Engelmann spruce and subalpine fir, 40-50% of dead, growing in bottom and up east side. Such salt and pepper mortality is characteristic of spruce beetle outbreak. Fire spot near top of ridge. Photo: Sudworth, Oct. 10, 1898 (319).

outbreaks (Schmid and Frye 1977), there is the potential that management can increase the incidence of SB outbreaks over its historic range of variability, certainly at stand and landscape scales. However, current management activities aim to mitigate this potential and are often focused on reducing small outbreaks. At a regional scale there is abundant evidence that extensive SB outbreaks occurred in western Colorado prior to the 20th century. If aregional scale SB outbreak were to affect the GM during the 21st century that would be considered within the range

of historical variability for this type of widespread but infrequent natural event.

Western balsam bark beetle: Western balsam bark beetle (*Dryocoetes confusus*) is known to cause mortality of subalpine fir in western North America (e.g. McMillin et al. 2003). The mortality can be caused directly by the beetles or by the associated blue stain fungus. Trees that are susceptible to attack generally have a lower percentage of the bole covered with constant crown, lower crown volume, lower radial

Plate 6—View of lake on head of Big Creek, looking south, shows Engelmann spruce and subalpine fir on north side of slide rock divide. The pattern of mortality is characteristic of spruce beetle outbreak. Photo: Sudworth, Oct. 13, 1898 (327).

growth in the last 5 years, and are older than trees that resist attack (Bleiker et al. 2003). In Colorado, *D. confusus* has been found in subalpine fir that were dying from root diseases (James and Goheen 1980). The current widespread mortality of subalpine fir in Colorado may be partly due to a combination of drought and western balsam bark beetle, but the relative influence of this beetle on mortality of fir is presently poorly understood. Currently, very little isknown about the HRV of Western balsam bark beetle. Therefore no assessment is possible of whether current trends are within HRV.

Montane woodland beetles: A number of bark beetles attack pinyon pine (Ronco 1990). The pinyon *Ips* (*Ips confusus*) is the most dominant beetle found in pinyon – juniper woodlands and it can cause severe mortality of pinyon pine (Wilson and Tkacz 1992, Leatherman and Kondratieff 2003). Trees may be predisposed to attack by Ips by black-stained root disease, dwarf mistletoe, and root damage (Leatherman and Kondratieff 2003). Outbreaks may be associated with extended drought that weakens trees' defenses to attack (Rogers 1993, Leatherman and Kondratieff 2003). Most often, host trees are killed by Ips attack. At a broader scale, mortality is often patchy, but
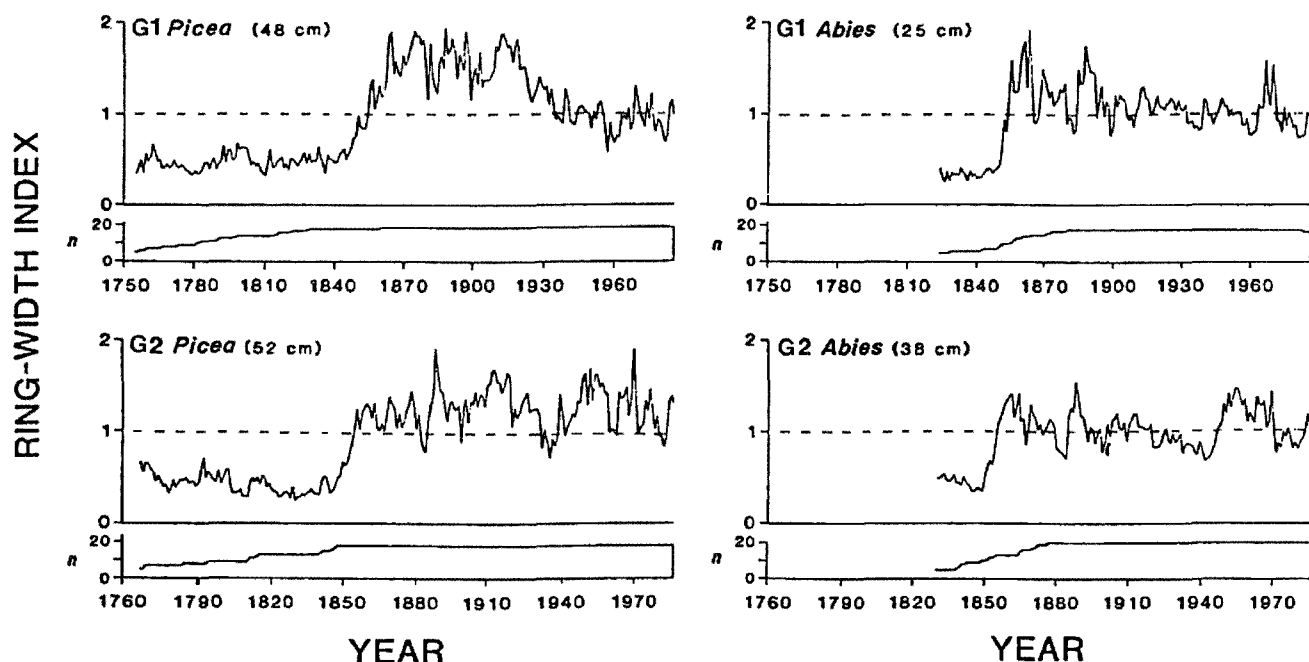
Figure 5.5—Mean ring-width indices for selected older trees in each stand show effect of spruce beetle outbreak in the 1850s and 1940s. The number in parentheses is the median diameter at breast height of trees sampled. G1 is Cottonwood Lake and G2 is Big Creek Reservoir. Figure from Veblen *et al.* 1991c.

extensive outbreaks over large contiguous areas can also occur. Outbreaks that cause mortality of pinyon favor the dominance of juniper within the stand. The bark beetles *Pityophthorus spp.* and *Pityogenes spp.* damage twigs and branches, particularly on shaded or storm damaged twigs and branches. Larger outbreaks of these latter two types of beetles can also occur (Rogers 1993). Several *Agrilus spp.* bark beetles burrow into the phloem of Gamble oak and sometimes cause mortality of individual stems (Furniss and Barr 1975).

The historic range of variability of these montane woodland beetles is not well understood in the GM area or in other areas that may be considered analogous. However, these beetles are native to the Rocky Mountain region and therefore their activity, at least at some level, is part of the natural landscape.

5.4.2. Defoliating Insects

Western Tent Caterpillar: The western tent caterpillar (WTC) is a native caterpillar of the Rocky Mountains that defoliates primarily aspen above 2400 m (7,870 ft; Schmid and Mata 1996). Endemic populations cause partial defoliation that rarely results in the death of trees over a limited spatial extent. However, during epidemics complete defoliation for multiple years may result in substantial growth reduction and mortality in aspen stands. If defoliation causes the death of many overstory trees, the stand may be converted to a stand of young aspen stems or succession towards dominance by shade-tolerant conifers may be accelerated.

Near La Veta Pass in the southern Front Range, stands suffered heavy mortality after 6 years of nearly complete defoliation by the WTC in 1980 (Schmid and Mata 1996). For the aspen stands of northern New Mexico and southern Colorado, the frequency of epidemics is 10 to 20 years, but single epidemics can

last more than 10 years. Stand structure does not appear to affect WTC populations, and thus any anthropogenic changes to aspen stands are not likely to alter the dynamics of WTC. Presently, there is no evidence that changes in stand structure or fire exclusion over the 20th century have influenced significantly the occurrence of tent caterpillar epidemics in the GM area or in other Colorado forests.

Butterfly looper: Several insect species use Gambel oak as hosts, but they rarely cause mortality of the oak (Harper *et al.* 1985). Among these insects are several species of aphids and wasps. Butterfly looper (*Lambdina punctata*) has been found to defoliate Gambel oak in Colorado (Brown 1958). Little is known about potential long-term changes in the ecological role of butterfly looper in western Colorado.

5.4.3. Pathogens

Dwarf Mistletoes: Dwarf mistletoes (*Arceuthobium* spp.) are hemi-parasites that infect most pine species and Douglas-fir in Colorado and can also affect juniper. Pinyon dwarf mistletoe (*A. divaricatum*) can cause severe damage and mortality to pinyon pine over limited areas (Rogers 1993, Gottfried *et al.* 1995). Mistletoe infections weaken trees and make them more susceptible to attack by other pathogens, such as bark beetles (Frye and Landis 1975). Although timber resources have been impacted severely in many areas, mistletoes are important for structuring forest stands and for creating habitat and resources for bird communities (Bennetts et al. 1996). While mistletoe infections are an important consideration where the primary management goal is timber production, deformed trees due to mistletoe infections are clearly part of the historic range of variability.

Macrofossils of dwarf mistletoes have been found in packrat middens as far back as 21,500 (± 500) years ago in the Southwest (Van Devender and Hawksworth 1986) which serves as evidence that these parasites and their hosts have co-evolved over long periods in this region. For the southern Front Range, Jack (1900:45) noted that mistletoe "was found to check and distort the growth of a great many trees in some localities, and in some cases it eventually caused their death." We are aware of no evidence of increases in mistletoe in the GM or surrounding areas. It is clear that mistletoe has been a natural part of ecosystems in the Colorado area for a long time.

Armillaria and Other Root Diseases: Root diseases caused by various species of decay fungi are common throughout conifer forests of the western US (Wood 1983). Decay fungi injure trees by killing roots or causing heart rots. *Armillaria* can be a direct agent of tree mortality and trees may also die because of increased susceptibility to bark beetles or windthrow. Fungi can persist for decades in the roots of stumps or snags and spread through root contact between healthy and infected trees (Dahms and Geils 1997). Mortality is usually of canopy trees, and disease centers may persist for hundreds of years (Dahms and Geils 1997). In Southwestern coniferous forests, it has been found that root diseases and their associated pathogens are responsible for mortality of 34% of trees in stands surveyed (Wood 1983). In southern Colorado, James and Goheen (1980) found that 80% of the trees infected with root disease also had bark beetles, suggesting that the root pathogens initially invade trees and predispose them to attack by insects. Dahms and Geils (1997) suggest there may be a positive feedback between fungi and beetles, with greater beetle-caused mortality resulting in greater sources for fungi inoculum.

There does not appear to be any historical information on root diseases for the GM area, although like other pathogens in conifer forests of the GM, these species are natives and have co-evolved with their hosts. For the Southwest, Dahms and Geils (1997) speculate that root disease fungi have responded to increased host material, especially the presence of stumps in many logged stands. Increased tree density also increases chances of root contact with infected trees in disease centers, causing further mortality and infection in stands. However, for the GM as a whole there is no regional evidence that current levels of root diseases are outside the historic range of variability.

Aspen Diseases: Aspen stands are often severely affected by a variety of diseases that cause tree mortality and damage the commercial value of the wood in Colorado (Hinds 1985). These include fungi and viruses that cause leaf diseases, many decay fungi that attack stems and roots, and canker-causing fungi that attack the bark. Such diseases are native to the southern Rockies, but at least at a stand scale human activities may increase rates of infection. For example, canker infection associated with wounding of trees can increase dramatically in managed stands (Walters et al. 1982). At a stand scale, many aspen diseases appear to

have been increased by wounding associated with logging, but at a regional scale conversion of older stands to younger stands may have reduced the occurrence of some diseases (Hinds 1985). Wounding by elk, deer, and other wildlife also facilitates the spread of many aspen diseases.

In a survey of trees wounded by campers in Colorado, Hinds (1976) found an incidence of canker infection double that of trees unaffected by campers. Blazing of trees by early Euro-Americans as well as Native Americans probably also affected the health of individual trees. Thus, there are numerous mechanisms through which humans can alter the spread and severity of aspen diseases. However, there is no evidence that current levels of aspen diseases are outside the historic range of variability.

White trunk rot (*Phellinus tremulae*) is another common aspen disease and can be responsible for severe and extensive aspen decay. Like the canker diseases, white trunk rot enters through wounds, thus its spread and severity can be altered through human activities. The impacts of this rot are usually more severe in older stands. Thus, a landscape of older aspen stands may have a higher degree of white trunk rot. However, there is no evidence that aspen diseases have become more widespread over the past century, nor that current levels of aspen diseases are outside the historic range of variability.

5.4.4. Insect Outbreaks and Climatic Variation

Weather profoundly affects the life cycles of insects as well as the capability of trees to respond to insect attacks, yet the effects of climatic variation on the occurrence of insect outbreaks are poorly understood (Swetnam and Lynch 1993, Logan et al. 1995). For example, mortality of bark beetle is increased by cold winters, and low temperatures are likely to be the major restriction on bark beetle outbreaks at high elevations in the Southern Rockies (Massey and Wygant 1954, Frye *et al.* 1974). Generally, warmer temperatures promote bark beetle outbreaks both through their favorable influence on the life cycle of the insect and drought-related declines in the tree's ability to withstand attack (Frye et al. 1974, Amman 1977). Extended drought that weakens trees' defenses may predispose pinyon stands to outbreaks of Ips (Rogers 1993, Leatherman and Kondratieff 2003). However, non-climatic factors related to stand structure also play such important roles that the

association of outbreaks with particular types of weather is difficult to verify quantitatively.

Although it has long been believed that drought predisposes Douglas-fir stands to outbreaks of Western Spruce Budworm (WSB; Cates and Alexander 1982), recent research from Colorado and New Mexico suggests that it is instead wet periods that favor outbreaks in this region. In northern New Mexico, 24 tree-ring records of outbreaks from 1690 to 1989 indicate a tendency for outbreaks to coincide with years of increased spring precipitation (Swetnam and Lynch 1993). Tree-ring records from the Colorado Front Range also show that WSB outbreaks are associated with moister years (Veblen et al. unpublished data). Although non-climatic changes in stand structures may also play a role in predisposing stands to outbreaks (Swetnam and Lynch 1993), these data indicate a significant role is also played by climatic variation. Due to this influence of climate on insect outbreaks, caution should be applied when attributing outbreaks to stand structural changes caused by humans. Furthermore, the effect of climate on insect outbreaks may become more important with global warming. Future global climate change is likely to intensify insect behavior and to result in a redistribution of insects, resulting in the infestation of new forest types (Logan et al. 2003).

6. Vegetation Changes

Local and regional distributions of tree species and forest types depend on complex interactions between abiotic factors (e.g., climate, soils, topography) and biotic processes (e.g., competition) which are often significantly altered by disturbances. Disturbances occur from both natural and human sources. In addition, humans may be modifying climate at both regional and local scales. In this chapter, we review general trends in vegetation changes that have occurred in the GM region in response to both long-term natural variability (e.g., climate change and long-term migrations of species) and human activities.

6.1. Long-term Natural Variability in Vegetation Patterns

Recent human impacts on vegetation patterns need to be viewed in the context of longer-term ecological changes such as those associated with Quaternary climatic fluctuations. Based on fossil insect evidence,

mean July temperatures in the southern Rockies during the late glacial maximum (LGM; c. 24,000 to 18,000 years before present) are estimated to have been 9 to 10° C cooler than modern temperatures (Elias 1996). Modern summer temperatures were reached as early as 12,000 years B.P., and warmer than modern summer temperatures prevailed from 10,000 to 3,500 years B.P. (Elias 1996). Fossil pollen evidence from the Southern Rockies suggests that alpine treelines were about 300 m (980 ft) higher than present during 9,000 to 4,500 years B.P. (Fall 1988). Pollen evidence also suggests regional cooling from 4,500 to 3,100 B.P., slight warming between 3,000 to 2,000 yr B.P., and then cooling during 1550 to 1850 A.D. (Fall 1988). These broad shifts in temperatures, and associated fluctuations in precipitation (generally wetter in warmer periods and drier in cold periods), were accompanied by significant changes in vegetation patterns in the Colorado Rockies. In some cases, slow dispersal of species into areas that are climatically suitable for them may have continued to the present (Betancourt 1990).

During the LGM, limber pine was a dominant component of the montane forests over much of the western US, in contrast to its often restricted range today (Betancourt 1990). At the same time during the full glacial, ponderosa pine was restricted in its occurrence, apparently to a few mountain ranges in southern Arizona and Mexico. At the end of the glacial period, dominance of these two species switched, and the range of ponderosa pine greatly expanded across much of the western US, although limber pine still dominates at low to middle elevations in portions of southern Idaho, Montana, and much of Wyoming where ponderosa pine generally is absent. Two hypotheses have been put forth for this switch: 1) a reduction in summer rainfall over the West during the LGM that favored limber pine over ponderosa pine at lower elevations; and 2) changes in fire regimes to high frequency, low severity patterns during the Holocene that have favored ponderosa pine life histories (Betancourt 1990).

A long-term (c. 8000 yr) study of fire and vegetation dynamics from a bog at 2920 m in western Colorado near Crested Butte documents the importance of historical and climatic factors in vegetation change (Fall 1997). Pollen and charcoal macrofossils were used to document changes in species dominance with changes in climate and in response to assumed changes in fire occurrence. From 8000 to 2600 years ago, the site was surrounded by a subalpine forest with greater dominance of Engelmann spruce than in nearby contemporary forests. Forest composition during this period implies a wet summer climate similar to that of the modern southern Rocky Mountains and Colorado Plateau. In some areas, aspen was successional to the spruce-fir forest during this period, although there also appears to have been a subalpine meadow or grassland present between 6400 and 4400 BP (Fall 1997).

On shorter time scales, other climatically-driven changes in vegetation patterns may be the result of lags in the response of species to warmer conditions during the Holocene (Cole 1985, Betancourt 1990, Allen et al. 1998, Feiler 1994). Migrational lags may have important implications for understanding biogeographical changes in the GM area, especially at lower elevations. The distribution of pinyon appears to have been moving north during recent millennia (Betancourt et al. 1991, Allen et al. 1998, Feiler 1994, Swetnam et al. 1999). The distribution of Gambel oak may also have been especially sensitive to climatic fluctuations. For example, the Gambel oak in the northern part of its range may have established in the mid-Holocene when summer rainfall was higher than today (Neilson and Wullstein 1986). While the distribution of oak appears to have been stable for centuries, based on aerial photograph interpretation in Utah, Gambel oak has been observed to be expanding its lower elevational range in the 20th century (Rogers 1982). Repeat photographs of the GM area also suggest that montane woodlands may be expanding their lower elevational extent (Bradford 1998). However, it is not clear whether this expansion is due to changes in climate, in the fire regime, grazing, or a combined influence.

Although the focus of this report is on vegetation changes related to human activities over the past few centuries, we include this brief consideration of longer term, natural changes in species distributions and community types to stress that vegetation flux is the norm. Thus, the challenge to ecologists and land managers is to discern how human activities modify the natural pattern of often slowly changing vegetation patterns. Species' responses to abrupt or gradual climate events must be taken into account when assessing changes that may be the result of human land-use changes. Abrupt climate change, such as a major drought or an anomalous cold period that causes widespread mortality of a species at its environmental limit, can be a cause of major shifts in ecotones (Allen

and Breshears 1998). Conversely, climate conditions favorable for plant regeneration may occur more slowly or episodically and lead to lags in re-establishment of an ecotone to some former position. Relatively subtle changes in climatic conditions can have long lasting impacts on vegetation patterns both through relatively direct effects on the recruitment and mortality of trees and indirectly through influences on disturbances such as fire and insect outbreaks. Tree-ring records show substantial annual-scale and decadal-scale climatic variation over the past 600 years in the southern Rocky Mountain region, including severe droughts lasting several decades in the late 1500s and the second half of the 19th century (Gray et al. 2003). Thus, there is a reasonably high probability that during the time period of increased human impacts (i.e. post-1850), vegetation conditions may have been in flux due to these fluctuations in the climate of the GM.

6.2. Human Influences on Vegetation Patterns

6.2.1. Impacts of Logging

Public concern over the effects of unrestricted harvesting, burning, and livestock grazing during the early settlement period were major reasons for establishment of the first Forest Reserves in Colorado, including those in the area of the GM (e.g., Jack 1900, Sudworth 1900, Ingwall 1923). Subsequently, timber harvesting has been one of the central foci of the management of these same areas and timber harvests on National Forest lands have continued to influence landscape patterns of forest structure and composition.

An exhaustive review of the complete ecological effects of logging is beyond the scope of this report, however, we will briefly highlight several of these impacts that may be relevant to the GM. The intensity of logging can vary from selective cutting, where few trees are removed, to moderate partial cuts, which significantly thin the stand, to clear-cuts of large areas. The intensity of logging will influence the associated ecological effects. For example, logging selected canopy tree species in the forest types found on the GM may change species composition, often towards more shade-tolerant species, such as subalpine fir. In contrast, clear-cuts favor the establishment of shade-intolerant species, such as aspen. Any such change in species composition and structure may affect future

forest dynamics, including susceptibility to subsequent natural disturbances. Logging in the Southern Rockies has been shown to have significantly affected landscape structure during the 20th century (Reed et al. 1996). Among other landscape-scale effects, the resultant fragmentation increased the number of patches and sum of edges while decreasing mean patch size. In addition to affecting landscape structure, logging has been shown to increase soil erosion, increase soil compaction and bulk density, and result in lower tree regeneration than following some natural disturbances, such as the 1997 blowdown in Routt National Forest in northwestern Colorado (C. Rumbaitis-del Rio *unpublished data*).

Logging, most obviously clear-cut logging but also selective logging, changes stand and/or landscape structure in a way that is potentially outside of the historic range of variability (Reed et al. 1996, Tinker et al. 1998). During the 20th century, most timber harvesting in Colorado has been concentrated in the spruce-fir, lodgepole pine, and aspen cover types of the subalpine zone. Clear-cutting was common from the 1950s to 1970s in the all these forest types in Colorado. Although the removal of the tree cover by intensive logging is superficially similar to some natural disturbances, there are important differences between the effects of logging and of disturbances such as intense fire, insect outbreaks, and blowdown. The degree of removal of organic matter from the site is much greater than in the case of natural disturbances, which leave abundant coarse woody debris on the site. Forest regeneration patches in previously logged areas typically lack the abundant dead-standing and fallen trees that are an important habitat for some wildlife species (Hutto 1995). Unlike fire, clear-cut logging does not create a blackened seedbed or remove all the fine litter. However, many clear-cuts and shelter-wood cuts are burned following logging to create a seedbed. Timber harvesting can also have a variety of impacts on soil properties (e.g. compaction) that are dissimilar to the impacts of fire on soils. As discussed below, logging and associated road construction fragment the landscape and create patterns that differ substantially from those created by natural disturbances. While impacts of early tree harvesting were profound in many forests of Colorado, the GM area was generally less affected, except for areas immediately surrounding coal mines, such as those in the upper valley of Coal Creek (Sudworth 1900; Plates 7 - 8). According to Sudworth's observations in 1898 (page 209), the

Plate 7—View of Coal Creek Canon looking west up creek, at a point 4-5 miles up from mouth in Gunnison National Forest, near GM. Douglas spruce and narrow leaf cottonwood with occasional small subalpine fir. Piles of ties in foreground. Photo: Sudworth, Sept. 28, 1898 (232).

timber in the GM was "exceedingly poor -- second- or third-rate, from which almost no clear lumber could be expected". There was relatively minimal timber harvesting by settlers in the 19th century, and of what timber was used, the dead timber was preferred because of its lighter weight.

Rocky Mountain Activities (RMACT; USDA Forest Service 2003 *unpublished data*) data on cover types affected by logging pertains mainly to harvesting during the second half of the 20th century (Figure 6.1; Table 6.1). These data indicate that spruce-fir is the

cover type with the largest percentage of surface area under some type of silvicultural management. Although 92% of the surface area of the GM is classified as not managed (Table 6.1) this reflects only the recent status of these sites and largely ignores early to mid 20th century harvesting activity that may have occurred. However, stand origin dates for the spruce-fir cover type (Figure 3.6) show that most stands originated in the mid- to late-19th century. This conclusion is also supported by more intensive dendrochronological data collected in the GM

Plate 8—View looking along timbered slope on south side of Coal Creek basin near coal mine where timber has been cut for timbering mine in Gunnison National Forest, near GM. Original growth Engelmann spruce and subalpine fir. Photo: Sudworth, Sept. 28, 1898 (238).

(Kulakowski and Veblen *unpublished data*). We interpret this to mean that most of the spruce-fir forest originated after 19th century fires, and that clear-cutting in the early part of the 20th century did not have a significant impact on current forest age structures. While past clear-cutting would have affected stand age, recent logging operations aim to retain a similar age structure of logged stands.

At lower elevations the most widespread woody vegetation types in the GM, oak shrublands and pinyon-juniper woodlands have been subject to relatively little cutting during recent years included in

the RMACT data base (Table 6.1). This contrasts with the norm in many pinyon-juniper woodlands in the Southern Rockies which have been subject to mechanical chaining, another form of active management, in an attempt to reduce woody plant cover (USDA Forest Service 1977, Grahame and Sisk 2002). Such management activities are ecologically quite different than natural disturbances such as fire or insect outbreak. Because mechanical chaining has an ecologically different effect from that of frequent fires and because such treatments are sometimes erroneously applied to stands that are not in need of
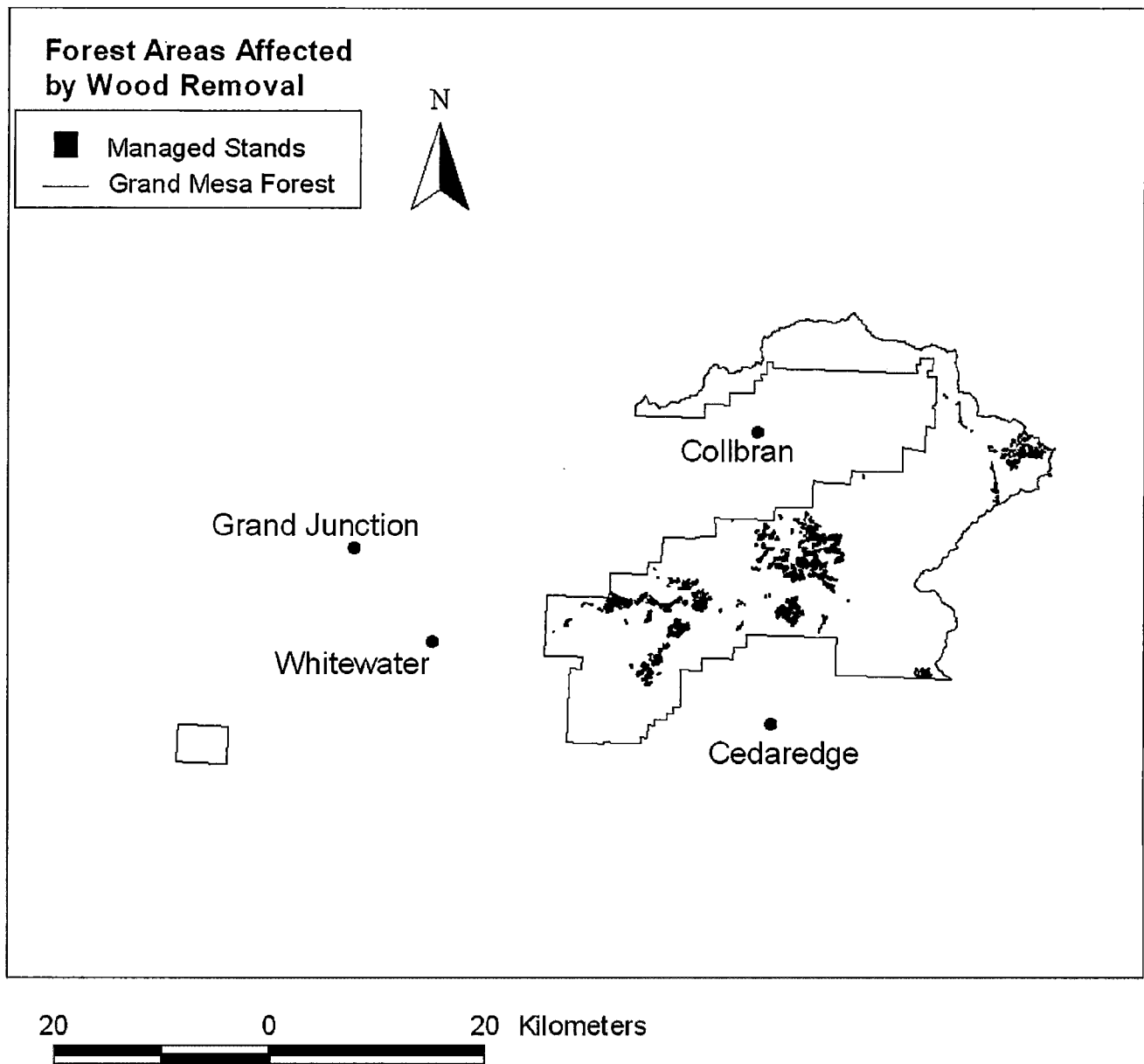
Figure 6.1—Map showing the location of managed stands in the GM.

restoration, chaining has had a detrimental effect on some nearby woodlands on the Uncompahgre Plateau (Eisenhart 2004). However, in recent decades cutting activities in pinyon-juniper woodlands in the GM apparently have been minimal.

6.2.2. Effects of Roads and Other Forms of Fragmentation

Landscape heterogeneity exists in natural ecosystems because of both underlying environmental variability and spatiotemporal variability in disturbance regimes (Knight and Reiners 2000, Veblen 2000). As an example, the wide-spread grass meadows on the

Table 6.1—Areas of forest cover types in the Grand Mesa National Forest affected by management activities. Source: USDA Forest Service RMACT database.

Cover Type	Total Area (ha)	Clear-cut (%)	Shelterwood (%)	Selective (%)	Salvage (%)	Unspecified Cutting (%)	Total Cutting (%)	Other Activity (%)
Aspen	48,642	1.17	0.06	0.00	0.00	0.20	1.43	1.88
Blue Spruce	22	0.00	0.00	0.00	0.00	0.00	0.00	55.23
Cottonwood	171	0.00	0.00	0.00	0.00	0.00	0.00	4.32
Douglas-fir	1,743	0.00	0.00	0.00	0.04	0.00	0.04	2.63
Forblands	5,733	0.08	0.16	0.00	0.13	0.06	0.44	3.98
Grasslands	13,739	0.06	0.19	0.01	0.03	4.82	5.11	1.69
Oak	24,403	0.00	0.00	0.00	0.00	2.71	2.71	10.53
Pinyon–Juniper	14,207	0.00	0.00	0.00	0.00	0.00	0.00	0.30
Ponderosa pine	14	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Sagebrush	1,421	0.01	0.00	0.00	0.00	2.33	2.34	5.90
Shrubland	6,126	0.00	0.43	0.00	0.70	0.20	1.32	1.07
Spruce–Fir	41,003	0.10	3.67	0.15	2.45	2.49	8.86	5.42
Willows	1,741	0.00	0.06	0.00	0.03	0.33	0.41	1.10
Total	158,965	0.39	1.01	0.04	0.67	1.57	3.68	4.05

GM result in considerable natural fragmentation of forests on the mesa. Superimposed on this heterogeneous landscape in the southern Rockies, human-caused alteration of natural patterns has occurred from reservoir and ditch construction, roads, fence lines, and power lines as well as tree harvest (Knight et al. 2000). Roads, fence lines, and power lines, as well as logging, are widespread causes of changes in landscape characteristics, typically fragmenting (breaking up into smaller parcels) formerly continuous habitats. The following discussion emphasizes the impact of roads and logging on fragmentation and ecosystem components. The Multiple Use Sustained Yield Act of 1960 establishes that National Forest Lands must be administered for a variety of purposes, including the production of various renewable resources. Discussion of the benefits of road and reservoir construction and logging to society is beyond the scope of this report, but such benefits obviously must be considered in making land-use decisions.

Roads may be among the most widespread impacts on natural landscapes (Diamondback 1990, Noss &Cooperrider 1994). Conversely, roadless areas (in addition to protected areas such as National Parks and Wilderness areas) may play an essential role in the movement and dispersal of species and the maintenance of ecological processes (e.g. Crist & Wilmer 2002). A complete consideration of the effects of roads is beyond the scope of this report, but reviews have recently been provided by Forman and Alexander (1998) and Trombulak and Frissell (2000). The magnitude of impacts of roads generally depends on the type and size of the road, the amount and type of use, the topography, climatic factors (length of growing season, precipitation, etc.), vegetation, and edaphic characteristics. Most obviously, the construction of roads kills sessile and slow-moving

organisms that are in the way of the road. Once the road is constructed, vehicles can cause mortality of animals by collision, which has been shown to have a significant impact on population of elk, lynx, owls, deer, and other animals in different ecosystems (Trombulak and Frissell 2000). While mortality generally increases with road traffic, some species are less likely to be killed on high-speed roads because the adjacent vegetation is often cleared, making it less attractive to some animals while increasing the visibility of animals near roads. Both paved and dirt roads also alter the behavior of animals. Roads can break up contiguous forest and, along with road use, can influence how animals such as elk, Northern goshawks, American martens, and others that are present on the GM use the forest (USDA Forest Service 2002). While multi-lane highways can clearly affect animal behavior for many species, narrow and unpaved roads can also alter the movement and range of some animals (*e.g.* Swihart and Slade 1984, Van Dyke et al. 1986, Baur and Baur 1990). In addition to impacting animals directly, roads result in soil compaction and other changes in soil properties, which can persist for decades after the road is no longer in use (e.g. Vora 1988). Roads can contribute to the spread of exotic species by altering the habitat in favor of non-native species and by increasing their dispersal by people. Roads may also facilitate other human impacts such as increased fire ignitions along roadsides, yet they also facilitate access for fire suppression activities.

Fragmentation by roads or logging of formerly continuous habitats, whether they be forested or other vegetation types, is recognized as a major threat worldwide to wildlife species (Wilcove 1985, Wilcove and May 1986, Knight et al. 2000). However, the direct impacts of habitat loss may be more important for some species (*e.g.* Hannon and Schmiegelow 2002, Schmiegelow and Monkkonen 2002, Lichstein et al. 2002). Forest fragmentation creates habitats that are less suitable for wildlife species that require forest interior habitats while also increasing habitat for species adapted to edges. Thus, forest fragmentation can have either positive or negative influences on the size of particular wildlife populations. However, forest fragmentation has the net effect of making rare species even rarer and common species more common (Beauvais 2000). The complete ecological effects of roads and logging on the ecosystems of the Rocky Mountains are not yet understood. Part of the

difficulty in accessing these effects resides in the fact that this landscape is inherently heterogeneous as a result of variations in biota (the mosaic of different forest types and meadows), topography, and substrate.

Forest fragmentation caused by road construction has been examined in western Colorado (McGarigal *et al.* 2001), north-central Colorado (Miller et al. 1996), Bighorn NF in north-central Wyoming (Tinker et al. 1998), and in the Medicine Bow NF in southeastern Wyoming (Reed et al. 1996). To assess the effects of clear-cut logging and road construction on landscape patterns, these studies have compared landscape patterns in roadless and roaded areas (Miller et al. 1996), utilized time series of RIS data from the 1950s to 1990s (Reed et al. 1996), and interpreted landscape configurations from natural disturbances *versus* logging and roads from satellite imagery (Tinker et al. 1998). These studies document major changes in landscape patterns associated with the impacts of roads and clear-cut logging. All four studies were conducted primarily in subalpine forests and their most important findings about the effects of clear-cutting and roads include: 1) decreases in patch sizes; 2) increases in patch densities, total edge perimeter, and edge densities; and 3) simplification of patch shapes at the landscape scale. The extensive network of reservoirs and ditches in the GM has also contributed to fragmentation and has altered natural ecosystems by creating new aquatic and lotic habitats and potentially fragmenting some of areas formerly continuous forest. It is possible that reservoirs and ditches today would have some effect on fire spread by serving as fire breaks.

Although natural processes (e.g., fire, blowdown) also can cause analogous changes, road construction and logging are generally different from natural patterns in terms of the microsites created as well as the landscape scale patterns that are created (Mladenoff et al. 1993). While research has been limited, current knowledge of habitat use in the Rocky Mountain region suggests that in some areas, clear-cutting and road construction are shifting mammalian populations toward more generalist and human-tolerant species at the expense of forest-adapted species (Beauvais 2000). While further work is necessary to consider landscape diversity analysis to account for differing levels of natural heterogeneity due to topography, soils, and rock, it appears certain that the types of habitats created by road construction are outside the historic range of variability of the landscapes of the southern

Rocky Mountains.

Roads vary in width, permeability, and frequency of use, and, consequently, all roads do not have the same ecological effect. Many of the roads of the GM are unpaved and are used by ATVs in summer and by snow mobiles in the winter. Unfortunately, adequate data on types of roads and overall use is not available. Hence, all roads were considered the same for the following analytical purposes. We measured over 1,800 km (1,120 miles) of roads in GMNF, in a geographic information system, for an overall road density of 1.16 km/km^2 (1.87 miles/mile^2 ; Figure 6.2). This density is high compared to other national forests
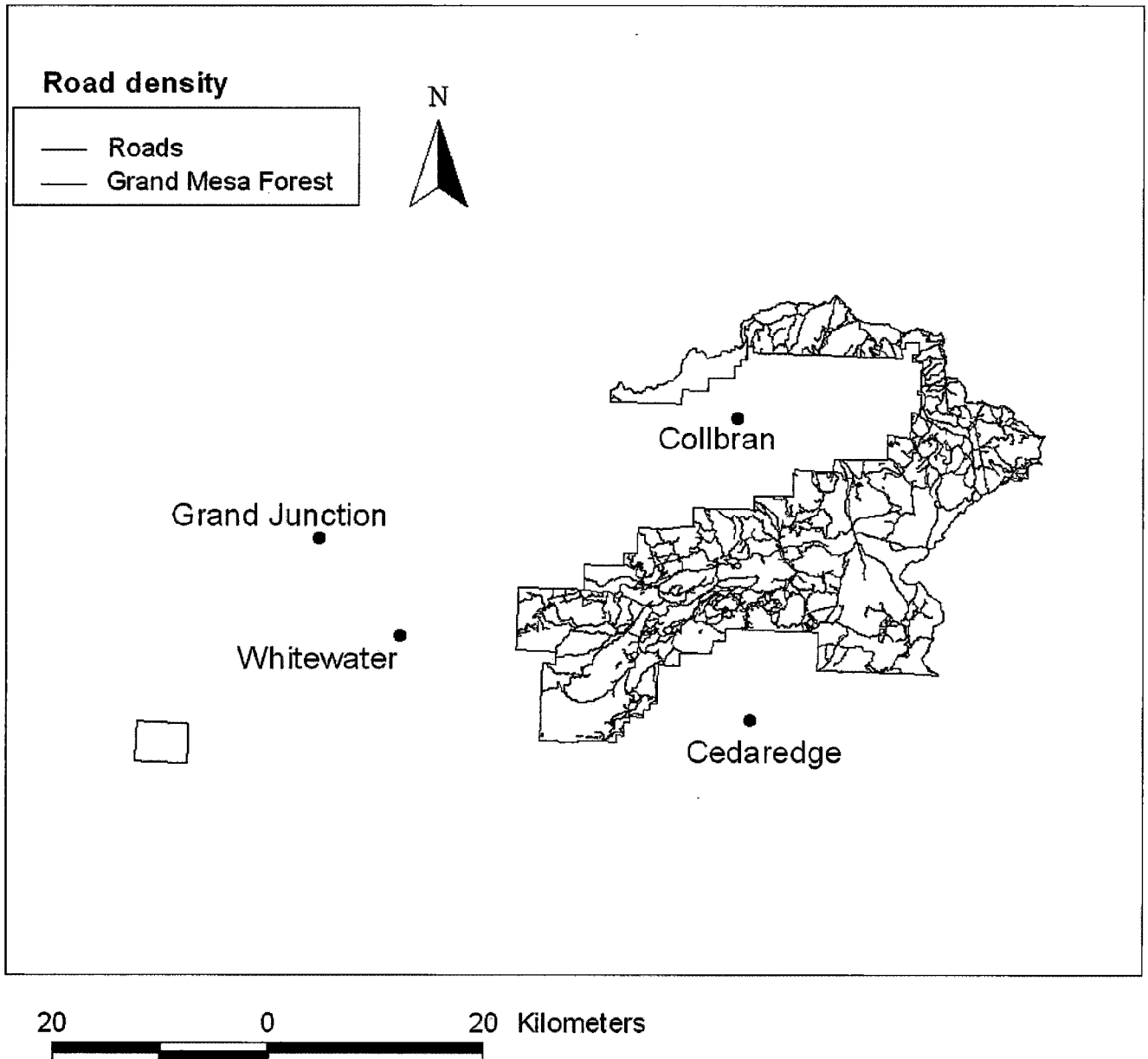

Figure 6.2—Map showing the location of roads in the GM.

in Colorado (e.g. Arapaho and Roosevelt NF) and, of course, this density varies considerably for different areas of the forest. No examination of the effects of roads on fragmentation, especially relative to the natural fragmentation associated with the widespread grass meadows, has been conducted on the GM.

In the GM, special consideration is given to the effects of roads on the threatened Canada lynx (*Lynx Canadensis*) (USDA Forest Service 2002). On the GM, lynx habitat is composed primarily of old spruce-fir forests and secondarily of mixed stands of aspen and conifers. Little is known about the effects of roads on lynx, but the construction of roads may reduce lynx habitat by removing forest cover and fragmenting the landscape. Because lynx avoid roads with high traffic volumes, the intensity of road use will influence the impact of roads on lynx. Motorized travel on the GM was permitted "anywhere" prior to 1994, but was then restricted to "designated routes only". This substantially curtailed motorcycle and ATV riding on the GM and reduced associated environmental impacts. As of 2002, there are 640 km (400 miles) of routes available for off highway vehicle (OHV) travel on the GM and 210 km (130 miles) of routes not open to motorized travel (USDA Forest Service 2002). Presently, a proposal is under consideration to reopen to motorized travel 100 miles of the 299 miles of non-system routes closed in 1994 (USDA Forest Service 2002). While no new roads would be constructed, such a decision would increase those effects on ecosystem function and wildlife that are related to intensity of road use.

6.2.3. Livestock Impacts

In addition to forest harvesting, grazing of livestock is one of the main uses of National Forest lands. Heavy grazing by livestock in many parts of the world has been associated with reduction or extirpation of some native plant species, proliferation of exotic plant species, and major changes in vegetation structure (Jarvis 1979, Huenneke 1988). Livestock change the vegetation through selective feeding habits and the differential ability of plants to withstand grazing, browsing, and trampling. The altered vegetation composition in turn alters litter characteristics, sometimes resulting in changes in decomposition rates, soil biotic activity, and nutrient cycling. Effects of livestock on forest and woodland ecosystems are

highly variable according to site-specific conditions of plant species palatability, soil conditions, type of livestock, the livestock grazing regime, and even climatic conditions that affect availability of alternative forage species.

By the late nineteenth century, livestock were relatively abundant on the GM (Sudworth 1900), and it is likely that they significantly affected most vegetation types of the GM. Studies conducted elsewhere in the western U.S., but in ecosystems similar to those of the GM, suggest that livestock populations probably had significant impacts in the GM in terms of species composition of some plant community types, the dynamics of ecotones between arboreal and non-arboreal communities, and fire regimes. For example, Gambel oak is not palatable to domestic animals (except to goats) and therefore dominance of oak appears to be favored by grazing because of the decline of herbaceous species (Harper *et al.* 1985). Thus, oak density may have increased above HRV level. Livestock grazing has also been suggested to decrease herbaceous cover and therefore reduce fire frequency in montane woodlands in the Southern Rockies (Gottfried et al. 1995). In many subalpine habitats in the western U.S., tree invasions of meadows have been linked to decreased pressure from livestock following periods of overgrazing (Dunwiddie 1977). In many ecosystems in the West (including subalpine ecosystems), heavy grazing during the 19th century is believed to have facilitated a major shift from native plants to exotic invaders (Baker 1978, Mack 1989, Dull 1999).

Overgrazing by livestock can reduce herbaceous ground cover, therefore altering the fire regime by reducing fuel continuity. The reduction in herbaceous cover can also reduce competition and therefore favor establishment and growth of pinyon and juniper (Gottfried *et al.* 1995). Overgrazing by livestock has been suggested as a possible explanation for increases in stand densities in montane woodlands in the Southern Rockies (Gottfried et al. 1995). Overgrazing may have also promoted tree encroachment into grasslands, both by reducing competition from herbaceous plants and by reducing herbaceous fuels for fire. Heavy grazing can reduce competition from grasses and expose bare mineral soil for tree seedling establishment. The hypothesis that overgrazing has contributed to an increased density of shrubs and small trees in montane woodlands in GM has not been

examined by rigorous studies, but it merits consideration as an alternative or contributory mechanism to explanations based on changes in fire regimes. Alternatively, it has also been hypothesized that encroachment of pinyon and juniper may be a result of a warming and drying trend in climate during the past 400 years, which has allowed pinyon to migrate north from Mexico (Swetnam *et al.* 1999).

Early reports of patterns of livestock use and their impacts on the vegetation suggest that to at least some degree livestock had altered vegetation conditions in the GM by the late 1800s. Grazing of cattle and, secondarily, of horses was carried out throughout the Battlement Mesa Reserve in the late 19th century. While exact numbers of livestock in the 19th century are not available for the GM area, the density of livestock in this area was reported to be greater than in other Forest Reserves, such as the White River and some individual cattlemen had as many as 4,000 head of cattle on the reserve (Sudworth 1900). The cattle were owned by settlers on adjacent land and residents of nearby and distant towns. Regarding the GM area, Sudworth (1900) wrote:

> The grazing of cattle and horses is carried on throughout this reserve.... Nearly all the settlers located on land adjacent to the reserve use some portion of the reserve as a range for cattle and horses. In addition to these settlers, parties residing in small towns in the region have large herds of cattle and horses in the reserve....
>
> The interior portions of the reserve are the regions most heavily grazed.... Almost all of the smaller creeks with their sources in the reserve are grazed by small herds of cattle. Through these valleys the broader slopes of the reserve are reached more or less by grazing animals.
>
> "Cow camps" and corrals are established in the larger grazing valleys, where cattlemen and cowboys rendezvous from May to November in the care of their herds. No fencing for holding cattle has been done in the reserve except near Thompson Creek and on the southwest arm of Grand Mesa....
>
> The most marked effects of cattle grazing are always in the vicinity of the water courses, where the animals congregate more and more as summer advances.

Historical photographs also document locally severe effects of livestock grazing around the time of Euro-American settlement in the GM area (Bradford 1998). In the 1940s, 30,000 head of cattle and 30,000 sheep were permitted in the GM. The number of livestock has declined and in 2002, 10,169 head of cattle no sheep were grazed in GM between June 15 and Oct 15 (USDA Forest Service *unpublished data*).

Large native herbivores, such as deer and elk, occurred in the GM in the 19th century (Sudworth 1900). This raises the question of whether livestock were simply occupying niches vacated by large native herbivores whose populations may have declined in the late 19th century. However, there are important differences in the patterns of disturbance created by native herbivores and livestock (Lauenroth and Milchunas 1989). For example, deer and elk can damage or kill small trees either through excessive browsing or through rubbing their antlers against the bark. In contrast, damage and mortality of tree seedlings from cattle results primarily from trampling rather than browsing (Kingery and Graham 1991). Although it is important to recognize that pre-settlement fluctuations in the populations of native herbivores must have had impacts on vegetation conditions of the GM, the nature of these impacts can only be conjectured. Cattle and native herbivores have the potential to affect the abundance and composition of herbaceous cover, including cheat grass, a fuel that could alter the fire regime in portions of the GM. Furthermore, herbivores have the potential to affect the dynamics of native and introduced herbaceous species. Thus, it is important that more research be conducted on the effects of large herbivores on the ecosystems in the GM.

6.2.4. Possible Vegetation Changes due to Changes in Fire Regimes

Montane woodland cover types: In the absence of fire history studies for GM it is difficult to be certain if the extent and severity of burning in the 20th century in pinyon-juniper woodlands and oak woodlands, and therefore the structure of these ecosystems, is within the HRV. The conclusions of studies conducted in nearby areas in Colorado imply that the fire regime of these woodlands is not greatly, if at all, outside its HRV (see section 5.3.1). Current knowledge of fire history and fire effects in PJ and oak woodlands in southwestern Colorado does not support the

widespread notion that 20th century fire suppression has resulted in atypical fuel accumulation or woody encroachment (Floyd et al. 2000, 2003, 2004, Eisenhart 2004). Specifically for these woodland types in GM, despite some uncertainty and the possibility of local exceptions, we tentatively conclude that fire suppression has not moved the fire regime greatly, if at all, outside the HRV.

Aspen: Broad generalizations have been made about the impacts of fire, logging, and ungulates on the relative abundance of the aspen cover type in the Rocky Mountain region (Johnson 1994, Kay 1997, Baker et al. 1997, Suzuki et al. 1999). It is important to consider the applicability of these generalizations to the GM. Forest inventory data from National Forests in Arizona and New Mexico indicate a decrease in the extent of the aspen cover type by 46% between 1962 and 1986 as conifers have taken their place (Johnson 1994). This decrease in aspen coverage has been attributed to fire exclusion (Johnson 1994) which allows conifers to successionally replace aspen. Although forest inventory data for the Southwest document a decline in the abundance of aspen during the late 20th-century, this may not be a suitable reference period for assessing HRV.

For the GM area, limited data are available on the structure and composition of the forests burned during the late 19th century period of increased fire (Kulakowski et al. 2004, Kulakowski and Veblen *unpublished data*). These data, which consist of the species, including the extent of aspen prior to the late 19th century fires, suggest that one effect of the extensive burning of the late 19th century was an increase in the extent of aspen in the landscape (Kulakowski et al. 2004). In many areas fires are believed to be essential in preventing conifers from successionally replacing aspen in many areas and therefore to the persistence of aspen in the landscape. Consequently, fire suppression has been suggested to cause aspen decline in the Southern Rockies (Brown and DeByle 1987, Kay 1997, Rogers 2002). However, it has also been argued that the total aspen forest cover has increased in areas near GM over the past century as part of an overall increase in forest cover following widespread forest destruction in the late 19th century (Manier and Laven 2002). In fact, comparison of an 1898 vegetation map (Figure 6.3) with a 1998 map of cover type distribution (Figure 6.4) indicates an increase in aspen extent has also occurred in the GM

since the late 19th century (Figure 6.5). The historical map produced in 1898 (Sudworth 1900) depicted the extent of fires in the late 19th century as well as the pre-disturbance vegetation. We compared the extent of aspen prior to the last major episode of burning in this area to the present extent (Kulakowski et al. 2004). Based on this comparison, a larger portion of the current landscape is dominated by quaking aspen relative to the late 19th century, a period of extensive burning in this area. During the 20th century, aspen was persistent over most of its extent, even in the absence of fire. Fires of the late 19th century also increased aspen cover in stands that were previously dominated by spruce and fir. The net effect of severe disturbances during and after the late 19th century has increased aspen cover relative to the period that preceded these disturbances.

Broad-scale stand-origin data of aspen stands also suggest that a large number of aspen stands initiated following the fires of the late 19th century (Figure 3.6). The total area where spruce and fir have replaced aspen in the GM area is small in comparison to the area where aspen has increased or is persistent. Aspen persisted or increased its extent primarily at lowerelevations, where site conditions appear to favor aspen over spruce and fir. The comparison of the 1898 vegetation map with the modern vegetation map clearly shows that the present extent of aspen in the Grand Mesa area certainly is not less than it was in the late 19th century (Kulakowski et al. 2004). Instead, there appears to be more aspen in the modern landscape than in the late 19th century. We were not able to study potential changes in stand densities in the aspen cover type. But, as the extent of aspen and the processes that shape this landscape appear to be within HRV, it is likely that stand structure is also within HRV.

Although aspen is successional to conifers in some areas, the present findings suggest that this fluctuation may be within the range of historical variation of the past several centuries. Furthermore, aspen appears to form persistent, self-replacing stands over most of its extent in the Grand Mesa area. These findings do not suggest that present aspen extent is below historical levels in this part of western Colorado, or that associated dynamics are outside of their historical range of variability. These findings also do not support the notion that aspen is threatened because of fire exclusion or other management activities during the 20th century. Instead, in portions of the western
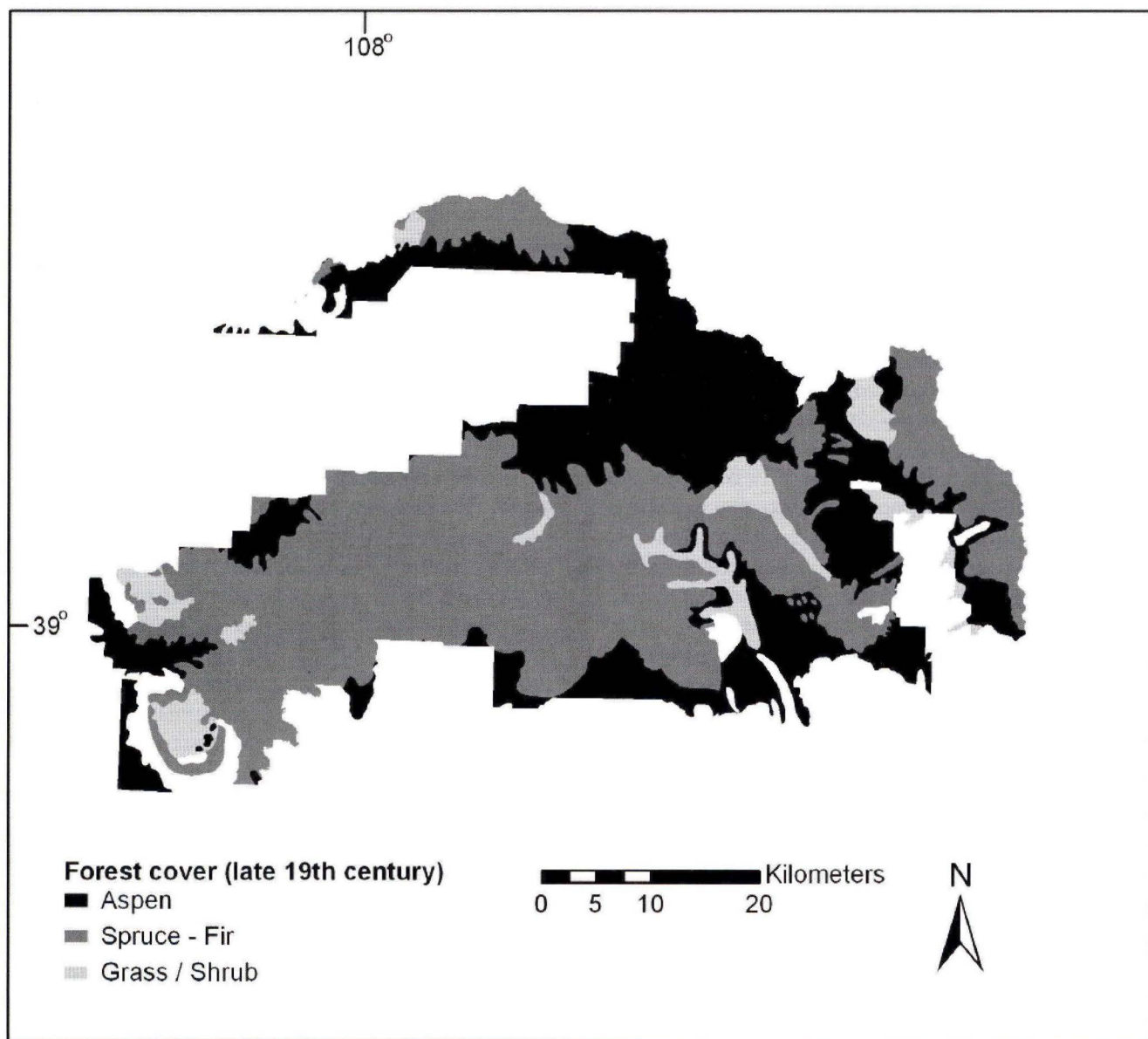
Figure 6.3—The extent of aspen, spruce–fir, and grass/shrubland dominance prior to the extensive burning of the late 19th century. Data are based on Sudworth (1900). Figure from Kulakowski et al.2004.

Colorado landscape the extent of aspen may varygreatly at a centennial scale in association with the infrequent natural disturbances that shape many of these forests. Where the successional replacement of aspen by conifers is occurring in such forests, such a trend may be within the range of historical variation.

Given the tendency for large areas of subalpine forest to burn in single years, it is unlikely that these elevated proportions of aspen are outside of the historic range of variability for this landscape. Nevertheless, in evaluating the probable decrease in the amount of aspen in this landscape during the late 20th and early 21st century, it is important to recognize that the extent of aspen was probably increased by the increase in burning of the late 19th century. Age structure of forests that were not burned in the late 19th century suggest that subalpine forests of western Colorado are characterized by even-sized and even-aged stands that
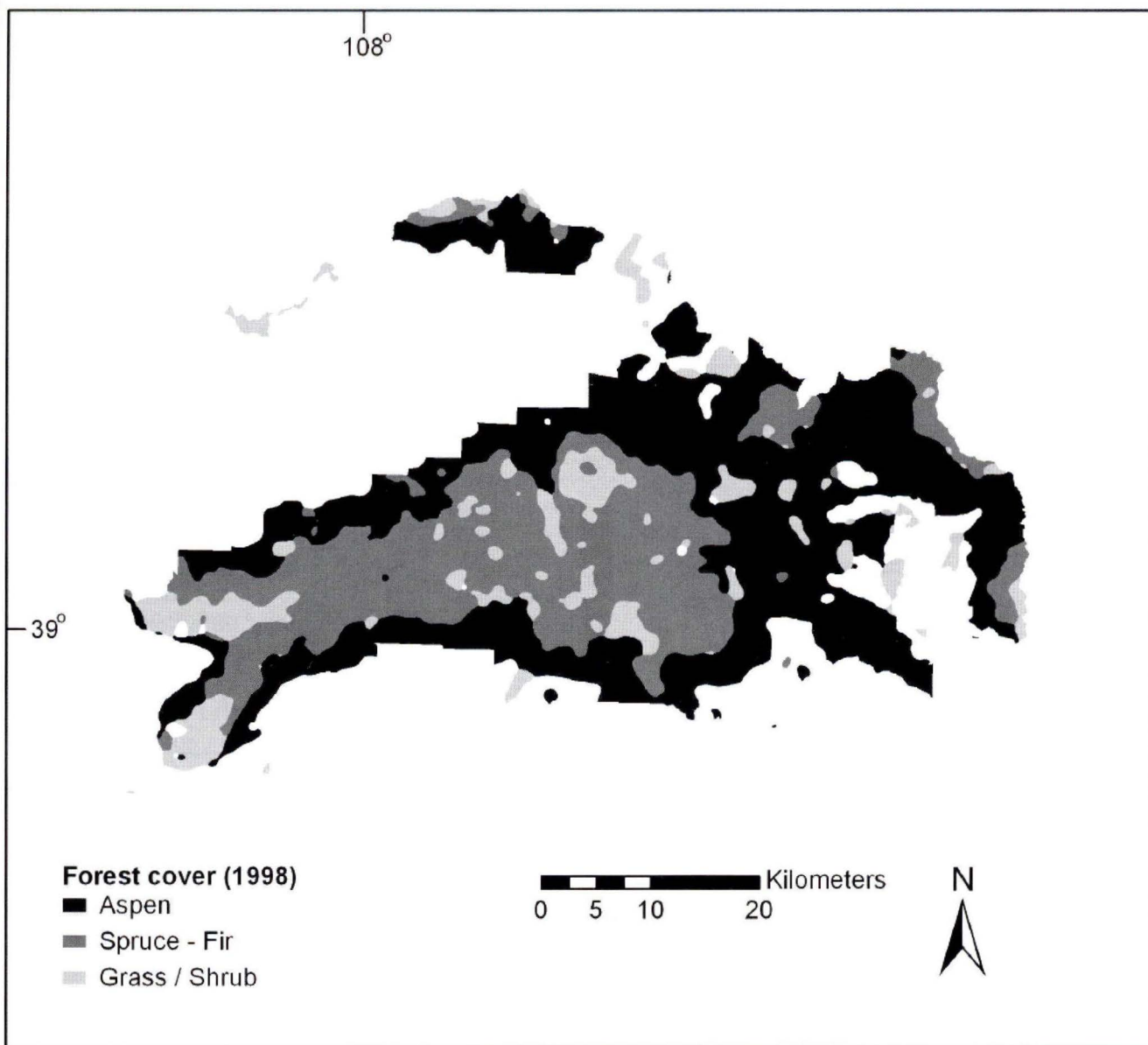
Figure 6.4—The extent of aspen, spruce–fir, and grass/shrubland dominance during the late 20th century. Data are based on USDA Forest Service (1998). Figure from Kulakowski et al.2004.

probably originated from earlier stand-replacing fires (Veblen at al 1994, Kulakowski and Veblen 2002, Kulakowski et al. 2003).

Since the late 19th century, comparison of historical and modern photographs shows that woody component of some plant communities is more prominent in some parts of the GM area (Bradford 1998). Bradford interpreted an increase of woody vegetation in pinyon-juniper, aspen, and spruce-fir cover types. However, the extent over which any such changes have occurred is not known. It is also not known whether any such changes are associated with fire regimes, climate, grazing, damming of rivers, or other processes. In forest areas that were burned in the late 19th century, the increase in vegetation is most likely a reestablishment and recovery following those
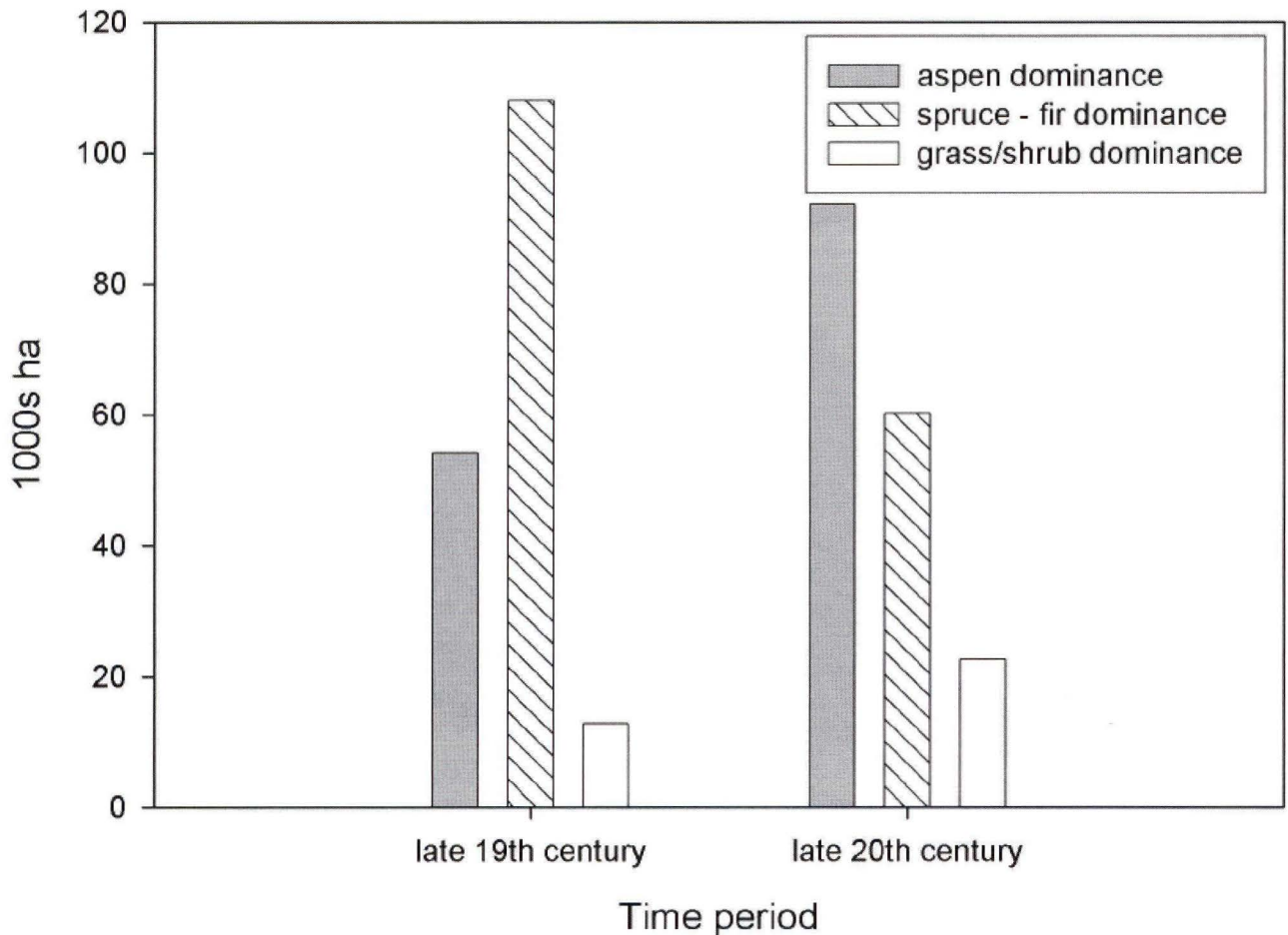
Figure 6.5—Comparison of cover dominance in the GM area by aspen, spruce-fir, and grass / shrub in the late 19th, prior to extensive burning (based on Sudworth 1900) versus the late 20th century (based on USDA Forest Service 1998). From Kulakowski et al. 2004.

disturbances.

Key to the notion that aspen has declined substantially due to reduced fire occurrence during the 20th century is the perception that aspen is usually seral to conifers. Certainly, at many sites aspen is seral to conifers where fire is excluded (Mueggler 1985, Peet 2000). However, in the GM the extent of sites where aspen is seral to conifers is less than the extent of sites where it can form self-maintaining stands (Kulakowski et al. 2004). Kay (1997) has argued that western aspen stands are "doomed" due to the combination of fire suppression and excessive browsing by large populations of native ungulates. His argument is that prior to the late 19th century, the Native American population was responsible for maintaining a high abundance of the aspen cover type through intentional burning and by maintaining low ungulate populations through hunting pressure. For the GM the lack of evidence of frequent fires in the subalpine zone is contradictory to Kay's argument. The second factor alleged by Kay (1997) to be "dooming" aspen is browsing by populations of ungulates. Because long-term data on elk and other herbivore populations are not available for the Grand Mesa, their effect on aspen dynamics over the past century is not known. However, the abundance of self-replacing aspen stands in the GM (Kulakowski et al. 2004) is a clear indication that this cover type is not in danger of

imminent demise due to elk browsing.

Engelmann spruce-subalpine fir cover type: As discussed in section 5.3.3, in general in the subalpine zone of the southern Rockies it is not certain that fire suppression has effectively reduced the area burned during the 20th century. Fire history studies conducted in nearby areas of spruce-fir forest in western Colorado indicate that the historic fire regime consisted of infrequent years of widespread burning that resulted in dense stands of Engelmann spruce and subalpine fir. The natural, long fire-free intervals in large areas of spruce-fir forests in the southern Rockies indicate that it is not logical to regard modern fire exclusion as resulting in unnatural fuel accumulations or unusual forest health conditions. Given the slow rates of tree growth and stand development in this habitat, it is unlikely that any reduction in fire occurrence in the 20th century has had an ecologically significant impact in the spruce-fir forests of the GM. While it is unknown how much stand initiation would have occurred without fire suppression, it is clear that the fire regime of the 20th century is not atypical when considering the long-term record. Furthermore, the widespread burning of subalpine forests in western Colorado in 2002 calls into question the effectiveness of fire suppression and also shows that stand-initiating disturbance (either fires or logging) has not ceased during the fire exclusion period.

7. Summary of Historic Range of Variability

In this final chapter we summarize the key findings about historic range of variability for each major forest cover type and identify some areas of urgently needed research. We also briefly discuss some possible management implications of our findings. Our intent is to illustrate where the existing level of knowledge is or is not adequate for informing possible management actions. We recognize that knowledge of historic range of variability is only one element in the management decision process (see, for example, the Committee of Scientists 1999). Although we use ecological restoration as an example of how historic range of variability can be incorporated into management, there are numerous other management goals (e.g., timber production, fire hazard mitigation, recreation) that would be included in the broader

planning process.

While some forest areas of the GM may have been substantially changed during the past century in comparison to reference period of the 17th to mid 19th centuries, other forest areas in the GM appear to be governed by processes that are within the historic range of variability. The changes that have occurred have resulted from a combination of human-causes and also climatic and other forms of natural variation. All of these factors together have contributed to both known and inferred changes in forest processes and patterns at stand, landscape, and regional scales. Consequently, separating the human-caused from the natural changes is difficult, and in many cases is impossible without substantial further research.

The general conclusion regarding HRV for the GM in many respects has resulted from a particular combination of circumstances unique to this location. These circumstances contrast with many other Rocky Mountain forests. Thus, conclusions regarding HRV and associated management may vary in their applicability to other Forests.

7.1. Summary for Major Forest Cover Types

Montane woodlands: No comprehensive study that compares reference and present conditions has been conducted in the montane pinyon-juniper and Gambel oak woodlands in GM. Studies from nearby areas of pinyon-juniper and oak woodlands (e.g. Gruell 1999, Floyd et al. 2000, 2004, Eisenhart 2004) are the principal sources of information on historic trends and current conditions in these woodlands. There appears to be a consistent pattern of reduced fire occurrence caused by fire suppression in montane woodlands during the early 20th century in areas near the GM (Gruell 1999, Floyd et al. 2000, 2004). However, following the period of active fire suppression and coincident with favorable fire climate, some montane woodlands appear to have returned to a fire regime that is not outside of the HRV (Floyd et al. 2000, 2004). Without research within the GM montane woodlands, it is not possible to determine whether the frequency of fire or area burned during the 20th century has been within or outside of the HRV. However, studies of nearby woodlands indicate that the historic fire regime was dominated by severe, stand-replacing fires rather than by non-lethal surface fires. Thus, any effects of 20th century fire suppression do not include a major shift in fire type from surface to stand-replacing fires

or an unprecedented accumulation of fuels. Even if the total area burned during the 20th century was reduced by fire suppression, the alteration of the fire regime has been relatively slight.

Similarly, relatively little is known about insect activity prior to and following Euro-American settlement. Important native populations of insects were active in montane woodlands of Colorado prior to Euro-American settlement. Thus, their presence and probable outbreaks are likely to be within the HRV for these woodlands, even though data are lacking on the extent and severity of pre-20th century insect outbreaks. The current outbreak of Ips beetles in PJ woodlands is interpreted as a response primarily to drought. Such outbreaks have probably occurred in association with pre-historic droughts, and there is no reason to attribute the outbreak to fire suppression or any other land-use practice.

Aspen cover type: The historic fire regime of the aspen cover type consisted of stand-replacing fires with a high degree of variance in the extent of burning at the scale of centuries due to the sensitivity of the subalpine fire regime to climatic variation. Although at some sites, aspen is seral to conifers and fire suppression would result in a decrease in aspen cover at those sites, there is no evidence that fire suppression in the 20th century has reduced the extent of aspen in the GM at a landscape or regional scale. At the majority of sites in the GM, aspen forms self-replacing stands even in the absence of disturbance (Kulakowski et al.2004) and its persistence in the landscape is not dependent on fire everywhere. Based on an historical map of vegetation patterns, the extent of aspen in the GM today appears to be greater than it was prior to the extensive burning of the late 19th century. Thus, where aspen is being replaced by conifers in the GM, such a trend may be a return to conditions more typical of the period that preceded the last major episode of fire in this area. The infrequent disturbances that shape aspen forests likely result in a broad range of conditions that can be considered within a range of natural variation. The extent of aspen in the GM today appears to be within the historic range of variability of the past several centuries.

Engelmann spruce-subalpine fir cover type : It is not certain if fire suppression activities have significantly altered fire regimes compared to the reference period. Widespread fires in this cover type are highly dependent on extreme weather conditions that occur infrequently. Given the long fire-free intervals typical of this cover type, it is unlikely that the 20th century fire regime has deviated significantly from the historic range of variability. At a century time scale, the extent of burning in the spruce-fir zone does not appear to be outside of the historic range of variability of the past several centuries. Similarly, given the dominance of the historic fire regime by stand-replacing fires there is no evidence to indicate a shift from predominantly non-lethal surface fires to an unprecedented potential for severe fires. Current fuel loads in spruce-fir forests appear to be well within their historic range of variability, and have historically supported severe, stand-replacing fires in the spruce-fir zone.

Analogously, there is no reason to believe that current levels of insect and pathogen infestation are outside the historic range of variability in the spruce-fir cover type. The spruce beetle outbreaks of the 20th century were preceded by documented severe outbreaks in the 19th century that occurred well before there could have been any significant impact on these forests by Euro-American resource-use practices. We stress that there is no evidence to support the view that fire suppression in the spruce-fir cover type has resulted in significant increases in pathogen or insect problems during the 20th century. The spruce-fir cover type is the cover type that has been most significantly affected by logging during the 20th century. However, logging in spruce-fir in GM forests has been less than in most Colorado spruce-fir forests. Logging may have reduced the extent of older stands but there is no evidence that at a landscape scale the logging has resulted in a significant departure in stand age distributions relative to the range expected for a landscape characterized by infrequent, stand-replacing fires.

7.2. Overview

The historical ecological patterns and processes described in this report can aid resource managers, planners and the general public in discussions of whether current trends in forest conditions are unprecedented or within the historic range of variability for the various ecosystem types. Departures from the historic range of these conditions can highlight areas that may require extra attention to and possibly a reconsideration of management decisions.

Changes in forest ecosystem conditions on the GM over the past century and a half can be attributed to both human and climatic influences. In the early settlement period (c. 1860 to 1910), localized timber harvest, possibly increased fire ignitions from mining and other settlement activities, and relatively intense livestock grazing affected parts of the GM. However, these changes have not been as significant as in other areas of the Southern Rockies such as the Front Range. Native Americans appear to have had relatively slight impacts on the GM, especially in the higher elevation habitats. Nevertheless, their influences on fire and game populations may have been ecologically significant at local scales and especially at lower elevations.

The late 19th and early 20th century period of reservoir and ditch construction had a landscape-scale effect on the GM. Clearly areas directly affected by such earthworks projects were severely affected, but the total surface area affected is a relatively moderate portion of the landscape. The overall impacts on forested ecosystems have probably been minor. Road construction has been a major source of landscape fragmentation during the 20th century which has probably had an ecologically important but largely undocumented impact on wildlife populations. In comparison with other subalpine ecosystems in Colorado, timber harvesting has had a relatively minor impact on the GM.,

Findings presented in this report are expected to be used as baseline components in an integrated Forest management planning process. The Committee of Scientists report (1999) suggests that sustainable ecosystems must be at the heart of any consideration of what constitutes sustainable economic and societal goals. Without a sustainable ecosystem, natural resources that provide both social and economic needs may be compromised in the future. In this report we have attempted to provide a baseline assessment of forest conditions of the Grand Mesa National Forest during the few centuries prior to Euro-American settlement and any associated human impacts. An understanding of the historic range of variation of ecosystem patterns and processes provide what may be the only objective model for sustainable ecosystem behavior and conditions that can be used for management planning and defining desired future conditions.

Some readers may be disappointed that the findings of this report do not provide a blue print for ecological restoration or other resource management activities. There is ample opportunity for restoration projects in relation to localized land-use practices (e.g. in relation to mining, reservoir construction, and road construction) as well as the pervasive influence of livestock. However, in relation to fire and insect outbreaks, we did not find evidence to support massive projects of ecological restoration in the GM. We found no evidence to support the notion that current forest conditions are significantly outside the historic range of variability due to human alterations of the historic fire regimes. This does not imply that localized projects on fuels reduction are not justified by other land-use considerations, but it does strongly imply that ecological restoration cannot be used as a justification for generalized forest thinning or other fuels reduction treatments.

7.3. Research needs

In this report, we have synthesized information from a variety of sources in order to assess the HRV of GM. While we have used local research whenever possible, there is a general scarcity of such work that has been conducted in the GM. Although research from other ecosystems is very useful, it cannot be uncritically accepted for the GM. In many cases, findings from elsewhere should be treated as hypotheses that need to be critically examined in the GM.

We stress the urgent need for studies of disturbance histories (fire, insect outbreaks) and their ecological impacts in order to inform ecosystem-based forest management in the GM. Such work should be conducted in all vegetation types in the GM. Below we list priority research needs for the GM.

- Further research is needed to understand the effects of climatic variation on the fire regimes of montane woodlands and subalpine forests in GM. Fires exert a dominant effect on the development of woodlands and forests in the GM, yet a specific understanding of how these fire regimes are influenced by climate is lacking.

- There is an urgent need for further research of fire regimes in aspen forests and their relationship to aspen persistence. An intensive field-based study of aspen age structure is needed in order to identify the abiotic habitat conditions under which aspen is

self-replacing as opposed to seral to conifers.

- In addition to fires, outbreaks of spruce beetle have been important in the development of subalpine forests in the GM. The history of spruce beetle outbreaks has been reconstructed from only a few sites in the GM, and there is a need to conduct similar reconstructions from an extensive network of sample sites to quantify the spatial extent and severity of past outbreaks

- Livestock grazing has been present on the GM since the 19th century. However, the cumulative consequences of grazing on woodlands and forests in the GM are in need of further study. For example, how have ungulate populations varied in densities over the past c. 150 years and to what extent are they capable of preventing the regeneration of aspen?

- Roads may represent a unique type of ecological disturbance that was not present in the GM area during the reference period. Studies specific to the GM are needed to evaluate the potential ecological impacts of roads on disturbance regimes.

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